



Effects of intake interruptions on dune infiltration systems in the Netherlands, their quantification and mitigation

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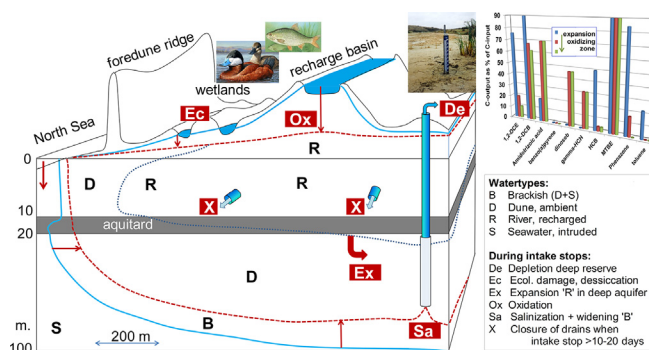
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HIGHLIGHTS

- Frequency and duration of intake stops will increase due to climate change.
- Hydrological, hydrochemical and ecological effects are serious but manageable.
- Effects of salt water intrusion and aquifer oxidation can be predicted.
- A magnitude scale for intake stops (MIS) is proposed.
- Presented mitigation strategies help to cope with intake interruptions.

GRAPHICAL ABSTRACT



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ABSTRACT

In the coastal dunes of the Western Netherlands, managed aquifer recharge (MAR) is applied for drinking water supply since 1957. The MAR systems belong to the Aquifer Transfer Recovery (ATR) type, because recharge and recovery are operated without interruption. This makes these systems very vulnerable to intake interruptions, which are expected to increase in frequency and duration due to climate change. Such interruptions are problematic, because: (i) groundwater recovery from dunes needs to continue to supply fresh drinking water to the Western Netherlands; (ii) risks of salt water intrusion are high, and (iii) MAR bordering wet dune slacks with an EU Natura 2000 status cannot survive for long without MAR.

In this paper, effects of intake stops are discussed and quantified. The hydrological effects consist of the decline of water tables, disappearance of flow-through dune lakes, reservoir depletion, salt water intrusion, disruption of rainwater lenses, and entrapped air hampering a rapid refill of the groundwater reservoir. Water quality effects include changes in (i) redox environment of the flushed aquifer, impacting the behavior of nutrients, calcium, sulfate and organic micro-pollutants, and (ii) the mixing ratio of water types. The main ecological impacts comprise the dying of organisms in recharge ponds and dune lakes, and a decline of biodiversity.

Effects of very long intake interruptions (years) are predicted via historical observations during the long overexploitation period (1900–1957) prior to MAR. A closed form analytical solution for safe yield of a semiconfined aquifer is proposed, together with a related upcoming risk index. Both also apply to the pumping from any fresh water lens without MAR.

Abbreviations: ASL, above sea level; ASR, aquifer storage recovery; ATR, aquifer transfer recovery; BAR, basin aquifer recharge; BSL, below sea level; DOC, dissolved organic carbon; MAR, managed aquifer recharge; MIS, magnitude of intake stop(s); MSL, mean sea level; MTBE, methyl tertiary butyl ether; OMP, organic micro pollutant; SAT, soil aquifer treatment; SDBI, safe depth to brackish interface.

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Some mitigation strategies are discussed, such as a dual intake, raising the storage capacity, earlier mud removal, and accelerated refilling of the reservoir. A magnitude scale for intake stops (MIS) is proposed.

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1. Introduction

Managed aquifer recharge (MAR) is increasingly applied in the world, in various ways, to restore groundwater tables, reverse salt water intrusion, store water for later use, improve the infiltrated surface water quality, or dispose of undesired water (Todd, 1959; Huisman and Olsthoorn, 1983; Asano, 1985; Dillon, 2005; Bouwer et al., 2008). The various MAR systems can be roughly subdivided into 2 systems (Chadha, 2002): direct (e.g. recharge basins and injection wells) and indirect (e.g. river bank filtration and groundwater dams).

Most direct systems are discontinuous, because either (i) the recharge is periodically stopped, due to lack of (good) water to infiltrate, freezing conditions in winter, or the necessity to frequently remove mud from infiltration basins, or (ii) the recovery is periodically stopped because the stored water is not needed. Examples of such systems are Soil Aquifer Treatment (SAT) and Aquifer Storage and Recovery (ASR). SAT systems are applied where sewage effluent or other wastewater is to be reused or polished up by aquifer passage (Fox et al., 2006). ASR systems aim at storing abundant water in the rainy season in an aquifer for later use during droughts (Pyne, 2005; Maliva and Missimer, 2010).

Aquifer Transfer Recovery (ATR) systems belong to the continuous, direct MAR category (Dillon, 2005). They consist of infiltration means (basins or wells), an aquifer to flush and store, and a remote recovery system composed of wells, drains or canals. They normally have a continuous supply of high quality infiltration water, a continuous water demand, and the need to use aquifer passage for water quality improvements.

Intake interruptions (stops) are and have always been a problem for MAR systems. For ATR systems they form a particular problem, because they are supposed to continuously deliver water after aquifer passage. This means that an intake interruption has to be coped with by one or a combination of the following options: (i) continuing the recharge via another intake if available; (ii) drawing water from the stored volume, (iii) drawing water from sources elsewhere, or (iv) reducing the delivery of drinking water.

In this contribution, we focus on dune infiltration systems in the Netherlands, which belong to the very vulnerable ATR type. Vulnerability to intake interruptions is high for 3 reasons. Firstly, the dune systems rely on surface water from the Rhine River or Meuse River, both of which are periodically heavily polluted (Aquapedia, 2014; Vliet and Zwolsman, 2008). Secondly, there is a high risk of intrusion of adjacent North Sea water and upconing of saline groundwater below the fresh groundwater lens, as experienced in the past (Stuyfzand, 1989b, 1993). And thirdly, the dune infiltration systems also support wet dune slacks with an EU Natura 2000 status. They contain protected flora and fauna, which cannot survive for long due to dependence on conditions sustained by MAR.

Climate change is expected to exacerbate the situation by increasing the frequency and duration of droughts and peak flows for both the Rhine River and Meuse River (Zwolsman and Van Bokhoven, 2007; Sjerps et al., 2017). Pollutants with a constant discharge will be less diluted during droughts, whereas water turbidity may become too high during peak flows. In addition, temperatures during summer droughts may become a problem due to a rise of air temperature and more effects of thermal pollution by e.g. nuclear power plants during low flow periods. The current water quality standards for water to infiltrate with the purpose to supply drinking water after recovery, as posed by the Dutch government (IB, 1993; see Table 5 for an incomplete listing), are of course crucial for intake stops.

The increasing risk of longer and more frequent intake interruptions for the Dutch dune infiltration systems triggered the study reported here. We believe that the Dutch situation is not unique in this respect, because all MAR systems either suffer from intake interruptions or thrive on them. The effects discussed or even quantified here, as well as the mitigating measures proposed, are therefore of general interest.

To date the effects of full intake stops have been very limited for all dune infiltration sites thanks to their relatively short duration, and the ability to abstract water from stored water or to continue infiltration with water from another source. Observations on effects are therefore mainly based on small scale infiltration stops (limited to one or a few recharge basins) in connection with mud removal or renovation. Effects of prolonged full intake interruptions without back-up by another source water are predicted by analytical solutions or extrapolation of historical observations during the long period of overexploitation (1900–1957) prior to MAR.

2. Dune infiltration sites

Currently, there are 8 operational dune infiltration systems in the coastal area of the Netherlands at the edge of the Rhine River delta, in the southeastern marginal part of the North Sea basin (Fig. 1 and Table 1). They contribute about 15% to a total annual production of 1100 Mm³ of drinking water in the Netherlands. The preparation of drinking water from surface water by dune infiltration involves pre-treatment near the intake, transport to the dunes (10–60 km), recharge by basins and recovery by wells, drains and canals after on average 70 m aquifer passage in 70 days, and post-treatment (Fig. 2). Growing water demands necessitated a steady increase in recharge and recollection. This was achieved by extension of recharge facilities at the expense of

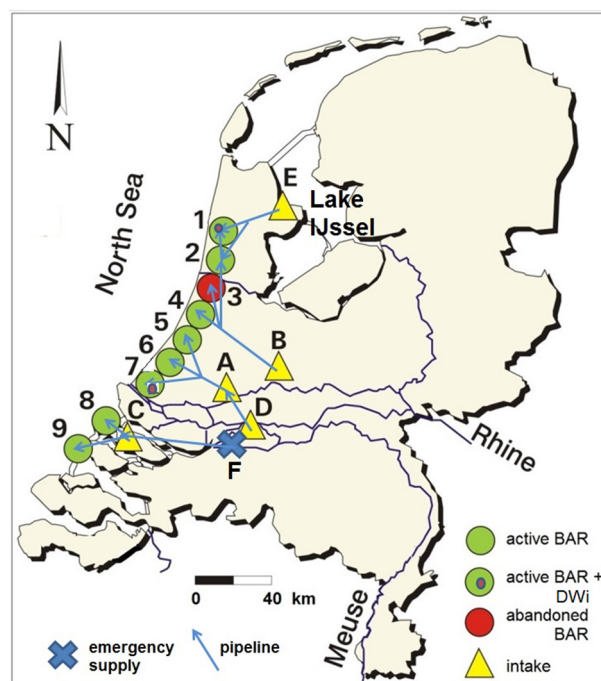


Fig. 1. Location of 8 operational and 1 abandoned basin aquifer recharge (BAR) production sites in the Netherlands, together with their surface water intake points. DWI = deep well injection. Further information in Table 1.

Table 1

Some details of BAR production sites 1–9 (No. 3 abandoned in 1999), with their surface water intake. DGE = dune groundwater extraction.

Site in Fig. 1			Start		Intake	Recharge		Pretreatment ^a	System ^b
No.	Near city	Name	DGE	BAR	Fig. 1	Source	Mm ³ /a		Inf/recovery
1	Castricum		1924	1957	E or (B)	Lake IJssel or Rhine R.	25	B + R + C	C + DWi / W
2	Wijk aan Zee	Kieftenvlak	1885	1975			17		C / W
3	Overveen	Groot Olmen	1898	1975–1999 †	B	Rhine R. or groundwater	1	R	B / W
4	Zandvoort	Leiduin	1853	1957			52		C / C + D
5	Katwijk	Berkheide	1878	1940	D or (A)	Meuse R. or Rhine R.	25	B + R	B / C + D + Q + W
6	Scheveningen	Meijendel	1874	1955			47		B + DWi / D + W
7	Monster	Solleveld	1887	1970			7		B + C / W
8	Ouddorp	Oostduinen	1934	1955	C or (F)	Rhine/Meuse estuary or Meuse R.	3.5	R	C / D + W
9	Haamstede		1930	1978			3.7		P / W

^a B = detention in basin or abandoned meander loop; C = activated carbon filtration + O₃ + UV; R = sedimentation or microfiltration + coagulation + RSF.^b B = basin; C = canal; D = drain; DWi = deep well injection; P = Pit; Q = horizontal well; W = vertical well.

natural dune valleys and by intensification of the process by changing from intermittent recharge to permanent recharge, by further pretreatment in order to reduce the clogging of basin floors, and by raising the gradient between recharge and recovery in various ways.

It took until 1990, when 2 ATR deep well injection systems were put in operation on sites 1 and 6. About 4 Mm³ of highly pretreated surface water is annually feeding about 20 recharge wells on each location. These systems are not considered in this paper.

On sites 1–9, geological formations to a depth of 100 m (in the south) and 450 m below sea level (BSL) in the north consist of unconsolidated sediments of Quaternary age (Jelgersma et al., 1970; Zagwijn, 1985). Above 2 to 5 m BSL, the deposits generally consist of dune sand on top of beach and shallow marine sands. Holocene aquitards composed of very fine silty sand, clay or peat, often somewhere between 7 and 20 m BSL, frequently separate the upper phreatic aquifer from the semi-confined second aquifer below (Stuyfzand, 1993). The fresh dune water lenses extend down to a maximum of 80–120 m BSL on all sites except for sites 2 and 7 where they reach 55 m BSL (Stuyfzand, 2017).

3. Intake stops

Intake stops can be characterized by their cause, duration, period (in winter, reservoirs are full, temperature and water demand low), time since previous intake interruption, anticipation time (announced or unforeseen), and response (e.g. reservoir depletion, tapping from another source, or restriction of drinking water delivery).

Total intake stops happened occasionally since the start of dune infiltration, due to freezing (becoming very rare), pollution peaks in the source water (by e.g. shipping accidents on river, industrial spills or low flows), problems with either the pretreatment facility or transport mains, or sea water intrusion via the river (during e.g. low flows and high sea level). A limited intake (or partial intake stop) means that basins in ecologically vulnerable areas may continue to operate, while others are drying up.

As an example, an overview of intake stops at pretreatment station B, in the period 1969–2015, is shown in Fig. 3. Since 1998 high quality groundwater is pumped from a well field next to the pretreatment facility, only when admixing with polluted Rhine River water helps to meet the quality standards or when a total intake stop can be temporarily buffered (the well field's annual production is limited by regulations to ~10% of the annual intake at station B).

Dune areas 1–2 and 5–7 can cope with intake stops by switching their source water from Lake IJssel (E) to Rhine River (B) or from Meuse (D) to Rhine River (A), respectively. Areas 8–9 can switch from estuarine Rhine + Meuse water (C) to Meuse River water (F). Thus, a dual anchor approach has been realized for each dune area, to raise the security of water supply. Nevertheless risks of a total intake stop remain, because both the primary and secondary intake may simultaneously suffer from the same pollutant, because intake B and E receive water from the Rhine River, and intake D, F and C from the Meuse River. There is, however, some time lag and for E and F a significant dilution.

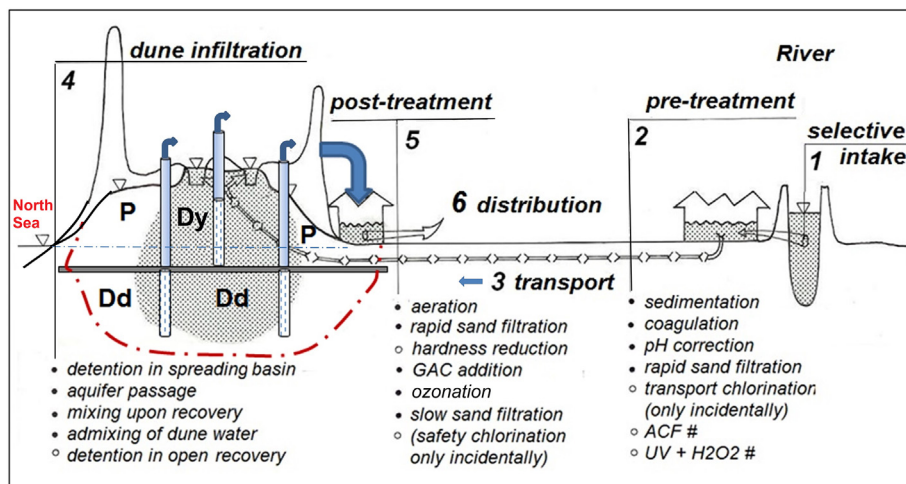


Fig. 2. General steps (from 1 to 6) in drinking water preparation through dune infiltration. Transport (3) over 57 km, dune area ~5 km wide, fresh water lens (red line) down to ~90 m below sea level. Open dots: not on all sites; # = Sites 1 + 2 only; ACF = activated carbon filtration; GAC granular activated carbon. Dy = dynamic reservoir (phreatic), mainly filled with infiltrated riverwater; P = phreatic dune reservoir, mainly filled with ambient dune groundwater; Dd = deep dune reservoir (semiconfined), filled with ambient dune groundwater and infiltrated riverwater. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

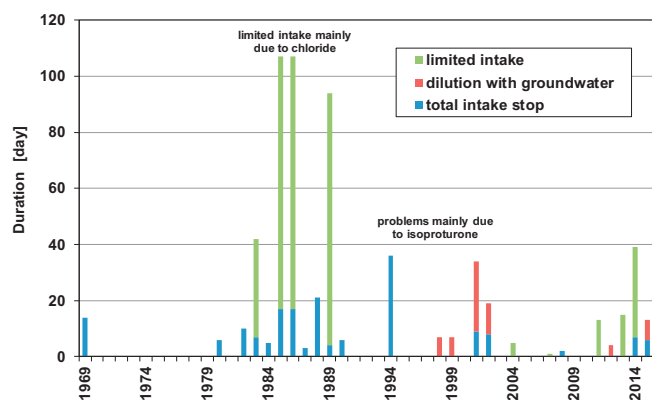


Fig. 3. Inventory of intake stops and periods with restricted intake or deliberate admixing of high quality groundwater, for pretreatment station B, which supplies infiltration water for sites 1–4 (currently mainly site 4). Data derived from RIWA-Rijn (2015).

The occurrence and duration of an intake interruption are often hard to predict, especially when an unknown pollutant has been discovered by a biological response at short distance. A better predictability holds for planned interruptions (for maintenance of the transport mains), for pollution peaks if detected and identified at an early stage far upstream of the intake point, and for low flow periods if developing during long term dry weather forecasts. Low flow periods have a predictable effect on water quality as long as all waste water discharges remain constant. In that case, the following equation (e.g. Davis and Zobrist, 1978) holds, assuming that the concentrations in river water are exclusively dictated by fluctuations in river flow:

$$C_X = \frac{a_X}{Q} + b_X \quad (1)$$

where: C_X = total concentration of constituent X in river water [mg/L]; a_X = average continuous anthropogenic discharge of X into river [g/s]; Q = river discharge [m^3/s]; b_X = (semi)natural background level of X in unfiltered sample [mg/L].

This relation can also be used to predict the water quality effects of future changes in river discharge, for instance due to climate change, as done by Zwolsman and Van Bokhoven (2007), and Sjerps et al. (2017).

The following scale is proposed to characterize the magnitude of an intake stop (MIS), with distinction between types A, B, C, D, and taking

into account a series of short interruptions (a, b, c, d etc.):

$$\text{MIS} = \text{rounddown}(1 + {}^2\log(n), 0), \text{ type} \quad (2)$$

with : $n = n_1 - a + n_2 - b + n_3 - c + n_4 - \dots$

where: MIS = Magnitude Intake Stops [–]; rounddown ($X, 0$) = function to rounddown X without decimal; type = seriousness of intake stop (Table 2); n = total number of days ($n_1 + n_2 + \dots n_X$) of one composite intake stop obeying the following conditions: the short time intervals a, b, c, d etc., during which water was taken in, should last <7 days, be shorter than half the previous and next stop, and each intake component should be of the same type (A, B, C or D).

As an example, we take 4 short stops of type C (total stop, dynamic reservoir available): $n_1 = 20$, $a = 8$, $n_2 = 6$, $b = 2$, $n_3 = 8$, $c = 9$, $n_4 = 7$, $d = 6$. This results in 3 separated intake stops of 20 days (MIS = 5C), $6 - 2 + 8 = 12$ days (MIS = 4C) and 7 days (MIS = 3C) respectively. The most severe intake interruption for dune infiltration systems in the period 1957–2016 was for dune areas 8–9 due to extreme Cl-concentrations of the Rhine River in combination with salt water intrusion via the Haringvliet sluices in the Rhine-Meuse estuary in 1985/1986 (MIS = 7C; Table 2).

For each dune infiltration area an emergency plan is under construction, indicating which water from which aquifer with which wells or drains is to be pumped out in a specific time order during a total intake interruption. The purpose is to ensure that declining water tables remain concentrated within the managed aquifer recharge production zones, to limit salt water intrusion and upconing, to maintain a high water quality of the water abstracted, and to safeguard the ecologically most valuable wet dune areas.

4. Bridging intake interruptions

In case of a total intake interruption, 3 reservoirs within the dune area may temporarily supply water (Fig. 2): the dynamic, phreatic dune and deep dune reservoir. The dynamic reservoir consists of the recharge basins, shallow recovery systems and the surrounding upper aquifer above 1–3 m above sea level, mainly filled with (infiltrated) surface water (normally there is no unsaturated zone under the basins). The phreatic dune reservoir surrounds the dynamic reservoir (also above sea level), and the deep dune reservoir consists of fresh groundwater, either ambient dune groundwater or infiltrated surface water, in the semiconfined aquifer(s), roughly beneath 10–20 m below sea level.

All emergency plans state that the dynamic reservoir is to be emptied first and subsequently the deep dune reservoir, while refraining

Table 2

Magnitude scale for intake interruptions, with estimated future frequency (2050), historical examples from dune infiltration sites in the Netherlands, and measures.

MIS	Duration [d]	Frequency [n/year]	Example (type)	Measures
1	<2	2	Repairs of transport mains (C)	Abstraction from dynamic reservoir, ecol. hotspots excluded
2	2–4	2	Phenol Rhine 2014 (C)	
3	4–8	3	Isoproturone Rhine 1999 (A)	
4	8–16	3	Endosulfan Rhine 1969 (C)	
5	16–32	1	chloride Rhine 1985 (C)	Abstraction from dune groundwater reservoir
6	32–64	0.25	Isoproturone Rhine 2001 (AC)	
7	64–128	0.1	Pyrazole 2015 Dunea (BC), chloride Haringvliet 1985/6 (C)	
8	128–256	0.01		Reduced abstraction from dune groundwater reservoir
9	256–512	0.005		
10	512–1024	0.001		
11	1024–2048	0.0001		
12	>2048	0.00001		
MIS-type:				
Type D				Total intake stop, dynamic reservoir not available (depleted or radioactive fallout)
Type C				Total intake stop, dynamic reservoir available
Type B				Total intake stop 1st anchor, full delivery by 2nd anchor
Type A				Partial intake stop 1st anchor, partial delivery by 2nd anchor

from abstraction from the phreatic reservoir. Important reasons are that (i) the natural, wet dune habitats are then least affected by the draw-down of water tables, and (ii) the switch to abstraction from the semiconfined (deep) aquifer takes time and has an impact on water quality with consequences for the post-treatment.

4.1. Using the dynamic reservoir

A dune infiltration system can be schematized as a single elongate recharge basin with bilateral recovery drains (Fig. 4), although on most sites various basins are aligned in parallel. The following 3 equations are derived from Huisman and Olsthoorn (1983), with addition of the forgotten term $w(\Delta H - D)$ in Eq. (5), to estimate the maximum bridging period from the dynamic reservoir ($t_{IP,1}$):

$$t_{IP,1} = \frac{V_{DYN,B} + V_{DYN,A}}{Q_{OUT}} \quad (3)$$

$$V_{DYN,B} = L w D \quad (4)$$

$$V_{DYN,A} = S_Y L (0.5 \Delta H 2B + w (\Delta H - D)) \quad (5)$$

where: B = horizontal distance between basin bank and drain [m]; w = width of recharge basin [m]; L = total length of recharge basin and drainage gallery [m]; D = mean water-filled depth of recharge basin [m]; Q_{OUT} = daily abstraction rate during the intake interruption, which is about equal to the input of surface water to infiltrate prior to the intake stop [m^3/d]; $V_{DYN,A}$, $V_{DYN,B}$ = amount of water in dynamic storage, within the aquifer (A) and basin (B) respectively, which can be used to cover a (short) intake stop [Mm^3]; t_{IP} = maximum bridging period from the dynamic reservoir which allows abstraction during an intake interruption [d]; S_Y = phreatic specific yield [–]; ΔH = normal maximum rise of the groundwater table above H_0 [m]; H_0 = original groundwater table or minimum level allowed during intake stop [m ASL].

The above given equations hold for a shallow phreatic aquifer, with following conditions: flow is assumed predominantly horizontal (Dupuit assumption), rainfall and evaporation cancel out, and all infiltrated water is recovered ($Q_{IN} = Q_{OUT}$).

The data for 7 dune infiltration areas in Fig. 4 lead to bridging periods with the dynamic reservoir, varying between 4 and 20 days, with more volume in the basins than in the aquifer for sites 5–7, more in the aquifer for sites 4 and 8, and about equal volumes for sites 1–2. Obviously, these volumes are not sufficient to cope with an unchanged water demand ($-Q_{IN}$ prior to the interruption) during more serious intake stops (MIS $\geq 5C$).

The bridging period can be extended by raising the water levels (ΔH) prior to a predicted intake interruption and by reducing the abstraction rate (Q_{OUT}). In case of nuclear fall-out, the dynamic reservoir will not be used!

4.2. Using the deep reservoir

In the period 1900–1956, deep dune groundwater abstraction from the semiconfined aquifer(s) led to lateral salt water intrusion from the North Sea and upconing of saline groundwater below the freshwater lens (Stuyfzand, 1993). These processes have been slowly reverted by MAR since ~1957, restoring most of the lost fresh groundwater reserve (Stuyfzand, 1993). Today this deep reservoir forms an important back-up for drinking water supply in case of intake stops and during peak demands. On average about 2–10% of the total water demand is currently supplied from the deep reservoir, without significant depletion because of recharge from the leaky dynamic and phreatic reservoirs.

The maximum bridging period from the deep reservoir ($t_{IP,2}$) is calculated by the following set of equations, assuming (i) no limitation in deep well pumping capacity, and (ii) a hemi-elliptical shape of the freshwater lens under the primary aquitard (c_1), with 2 safety zones (Fig. 5):

$$t_{IP,2} = \frac{V_{SAFE}}{Q_{OUT}} \quad (6)$$

$$V_{SAFE} = 0.25 \pi n_2 W B_2 \left(\frac{Z_1^2 - Z_0^2}{Z_2} \right) \quad (7)$$

$$\text{with: } Z_1 = Z_2 - D_{SAFE} \quad (8)$$

$$D_{SAFE} = f_1 f_2 \Delta Z \quad (9)$$

$$f_1 = \sqrt{\frac{1 + 0.001(c_1 - 500)}{1 + 0.001(c_2 - 500)}} \quad (10)$$

$$f_2 = \frac{Q_{OUT}}{K_F D_F n_W} \quad (11)$$

where: V_{SAFE} = maximum volume of fresh groundwater that can be safely pumped from the deep reservoir in case of an emergency [m^3]; Q_{OUT} = pumping rate from deep reservoir [m^3/d]; n_2 = effective porosity of deep reservoir [–]; W = length of coastal dune strip containing water catchment area [m]; B_2 = width of dune strip (from mean tide line to inland border of dune area) [m]; Z_0 = mean depth to base of well screens in deep reservoir [m BAT = below upper aquitard]; Z_2 = maximum depth in cross section to fresh/brackish interface, defined

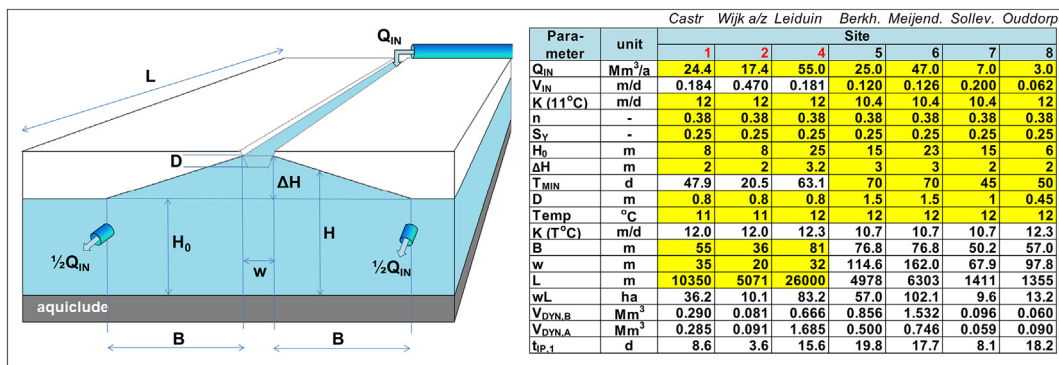


Fig. 4. Screen dump of a spreadsheet model to design a basin recharge scheme and to calculate the maximum bridging period ($t_{IP,1}$) of a total intake interruption utilizing the dynamic reservoir (water volume above H_0). Yellow cells = input. White cells = text or output. Input and calculations for sites 1–4 differ from those for sites 5–8 (site numbers correspond with Table 1 and Fig. 1). Explanation of most parameters in left panel and below Eqs. (3)–(5). In addition: V_{IN} = infiltration intensity; K = horizontal conductivity; n = porosity; T_{MIN} = minimum travel time in aquifer. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

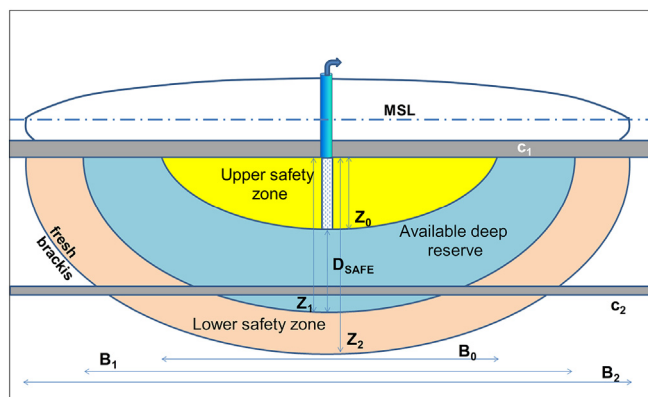


Fig. 5. Schematic of the deep reservoir with its upper and lower safety zone and the remaining deep reserve which is available during intake stops and peak demand. D_{SAFE} = thickness lower safety zone; c_1 , c_2 = upper and lower aquitard crossing the freshwater lens; MSL = mean sea level.

as 150 mg Cl/L [m BAT]; D_{SAFE} = thickness of lower vertical buffer/safety zone [m], ΔZ = uncorrected thickness of lower vertical buffer/safety zone, with a preset value based on experience, e.g. 30 [m]; f_1 = correction factor to account for leakage through aquitard 1 (low resistance means more recharge from above which reduces risk of salinization), and through aquitard 2 (low resistance means more recharge from below, which raises risk of salinization) [–]; c_1 , c_2 = vertical resistance to flow by aquitard 1 (above deep well screens), and aquitard 2 (below deep well screens) [d]; f_2 = correction factor to account for overpumping as indicated by the pumping rate compared to the transmissivity (KD) of the aquifer [–]; K_F = hydraulic conductivity of deep reservoir above fresh/salt interface [m/d]; D_F = ditto thickness [m]; n_W = number of pumping wells tapping deep reservoir [–].

In this approach, the safe freshwater volume is limited to Cl < 150 mg/L, which is the Dutch drinking water standard. The maximum depth in cross section to the fresh/brackish interface (Z_2) is to be estimated on Cl data from available deep piezometer nests, geophysical measurements or closed form analytical calculations (Stuyfzand, 2017). Piezometer nests are numerous in the coastal dunes of the Netherlands, each with 4–8 short (1–2 m long) well screens at different depth levels extending into the saline groundwater below the freshwater lens.

With an upper and lower safety zone, 3 hemi-ellipses are defined with maximum depths Z_0 , Z_1 and Z_2 respectively (Fig. 5). The upper safety zone is defined by Z_0 , and the lower zone by Z_1 and Z_2 . The different depth levels directly translate into a different width (B) because of

their linear interdependence if all other parameters remain constant ($B_0 = Z_0 / Z_2 B_2$ and $B_1 = Z_1 / Z_2 B_2$).

Application to 3 large dune infiltration sites (1 + 2 combined) yields the results presented in Table 3. The deep reservoir offers a large reserve which could supply water during an emergency for 4.5–6.5 years with low risks of salinization. The lower vertical buffer/safety zone (D_{SAFE}) is thinnest for sites 1 + 2 thanks to the advantageous resistance of aquitards (low c_1 and high c_2).

The above approach does not take into account the net effect of natural groundwater recharge of the deep reservoir (covering 2–10% of the water demand) nor the volumes of deep fresh groundwater within the tongues below the North Sea and, more importantly, further inland. The latter mainly formed by inland flow from the main dune area towards the reclaimed lakes or deep polders situated 3–5 km inland (Stuyfzand, 1993, 1995a). This means that more reserves are available in case of necessity, but this requires new wells to be drilled in the area in between the dunes and deep polders. Several waterworks need to drill new wells into the confined aquifer anyhow, if they wish to utilize the deep reserve at full capacity, because the current well capacity is not sufficient.

On the other hand, the calculated maximum bridging period from the deep reservoir may be unrealistic in case of either a too strong drawdown of the water table or hydraulic heads in the deep reservoir, or upconing of saline groundwater in wells at unfavorable locations (deviating from the schematized cross section in Fig. 5, due to heterogeneities). This calls for a check on the hydrological effects (next section).

5. Hydrological effects of reservoir depletion

5.1. Declining water tables

Drawing water from either the dynamic or deep dune reservoir will lead to a drawdown of hydraulic heads in the phreatic and semiconfined aquifers. This drawdown cannot be easily predicted because of (i) the complex surface water groundwater interaction in which also the presence of more or less muds plays a role; (ii) spatial variations in basin depth, basin shape, distance between basin banks and recovery, presence and hydraulic resistance of aquitards and mud deposits; (iii) temporal variations in water temperature (viscosity), clogging by muds, rainfall, evaporation, recharge and recovery rate; and (iv) slow changes in the position of the freshwater lens in response to long-term changes in recharge and pumping.

All these complications can be taken into account with a 3D transient flow model (Nienhuis et al., 2012). However, such a model is extremely data hungry and requires excessive running time, while probably being not very accurate. Therefore a simple analytical approach is proposed to

Table 3

Left panel: input parameters for calculating the maximum volume of fresh groundwater that can be safely pumped from the deep reservoir in case of an emergency (V_{SAFE}) and the maximum bridging period from the deep reservoir ($t_{IP,2}$) for dune infiltration sites 1 + 2, 4 and 6 (numbers corresponding with Figs. 1 and 4). Right panel: calculated results. BSL = below sea level; BAT = below aquitard 1.

Input data					Model output data				
Dune inf. area		1 + 2	4	6	Dune inf. area		1 + 2	4	6
Q _{OUT}	Mm ³ /a	41.8	55.0	47.0	B ₀	m	652	1195	1400
n ₂	–	0.35	0.35	0.35	B ₁	m	2863	2810	2740
B ₂	m	3180	4500	3500	Z ₁	m BAT	70.2	40.0	58.7
W	m	5100	10,000	7000	D _{SAFE}	m	7.8	24.0	16.3
Base aquitard 1	m BSL	20	18	15	f ₁	–	0.41	1.91	0.82
Top aquitard 2	m BSL	95	65	50	f ₂	–	0.63	0.42	0.67
Z ₀	m BAT	16	17	30	V ₀	m ³ /m	2869	5586	11,545
Z ₂ (150 mg Cl/L)	m BAT	78	64	75	V ₁	m ³ /m	55,270	30,881	44,218
Z ₃ (8500 mg Cl/L)	m BAT	90	79	110	V ₂	m ³ /m	68,184	79,168	72,158
ΔZ	m	30	30	30	V _{SAFE}	m ³ /m	52,401	25,295	32,673
c ₁	d	150	5000	500	V _{SAFE}	Mm ³	267	253	229
c ₂	d	3380	1000	1000	V ₂	Mm ³	348	792	505
K _F D _F	m ² /d	1755	1500	1125	V ₃	Mm ³	401	977	741
n _W	–	103	240	172	t _{IP,2}	year	6.4	4.6	4.9

predict the drawdown in areas of specific concern, e.g. where ecological damage should be avoided. The approach consists of 2 parts, if pumping from the dynamic reservoir continues beyond the threshold level.

The first part consists of a more or less trapezoidal initial groundwater elevation between recharge basin and bilateral recovery (Fig. 6). With the schematization of an entire dune infiltration system as presented in Fig. 4, the drawdown curve of water tables in the middle of the dynamic reservoir (at $X = B + w/2$), can be roughly estimated using the following water balance equations based on Fig. 6:

$$\text{if basin filled : } dH_B = \frac{fQ_{OUT}}{L(S_Y B + w)} \quad (12)$$

$$\text{if basin dry : } dH_A = \frac{fQ_{OUT}}{L S_Y (B + 0.5w)} \quad (13)$$

$$\text{with : } f = \frac{\left(\frac{D}{dH_B} + \frac{\Delta H - D}{dH_A} \right)}{t_{IP}} \quad (14)$$

where: dH_B , dH_A = daily drawdown of water table, in the basin when still holding water (b in Fig. 6) and under the basin when emptied (a in Fig. 6) [m/d]; Q_{OUT} = daily abstraction from dynamic reservoir [m^3/d]; f = correction factor to compensate for minor geometric errors [–].

The drawdown calculation of part 1 requires at least 2 steps, with $f = 1$ in the first step, followed by determination of f via Eq. (14) in step 2. This correction term is small (0.9–1.1) for the 3 cases elaborated in the inset of Fig. 6, depicting the drawdown for sites 1, 4 and 6. On site 6, the larger volume stored in basins explains the slow drawdown during the first 14.5 days. The drawdown is 2–4 times faster on all sites once the basins are dry, because of the porosity effect.

These equations permit to calculate the maximum drawdown (S_{DYN}) as function of time (Fig. 6), as follows:

$$\text{if } t \leq t_B \text{ (water in basin) : } S_{DYN} = t dH_B \quad (15A)$$

$$\text{if } t_B < t \leq t_{IP,1} \text{ (basin empty) : } S_{DYN} = (t - t_B) dH_A + t_B dH_B \quad (15B)$$

where: t = time since start of intake interruption [d]; t_B = time needed to empty basin = D/dH_B [d].

The position of the water table ($\varphi_{X,t}$) then becomes, as function of X (distance from e.g. a ditch, drain or row of wells) and time:

$$\varphi(X, t) = \varphi_0 + \frac{X'}{B'} (\varphi_B - \varphi_0 - S_{DYN}) \quad (16)$$

where:

φ_0 , φ_B = water table at $t = 0$, at $X = 0$ and $X = B$, respectively [m ASL].

for $0 \leq X \leq B$: $B' = B$ and $X' = X$

if $t \leq t_B$ (water in basin): for $B < X \leq B + w$: $X' = B$

for $B + w < X \leq 2B + w$: $X' = 2B + w - X$

if $t_B < t \leq t_{IP,1}$ (no water in basin):

for $X \leq B + w/2$: $B' = B + w/2$ and $X' = X$

for $B + w/2 < X \leq 2B + w$: $X' = 2B + w/2 - X$

The presented schematization yields a very rough approximation for the drawdown under the middle of the basins (via Eqs. (12)–(14)) and for the sinking position of the whole water table. The approach does not take into account the decreasing width of basins with decreasing depth, effects of basin clogging, the drawdown near the recovery system, and it assumes a linear water table gradient (which roughly corresponds with measurements). We need the second part to account for a further drawdown if pumping continues beyond the dynamic reservoir threshold.

The second part consists of Edelman's case B (Huisman and Olsthoorn, 1983; Bruggeman, 1999). This yields the time dependent drawdown of the groundwater table at distance X from e.g. a ditch, drain or row of wells ($S_{X,t}$), in case of a sudden start of constant unilateral drainage with q_0 [m^2/d ; negative for drainage]:

$$S_{X,t} = 2q_0 \left\{ -u \operatorname{erfc}(u) + \frac{1}{\sqrt{\pi}} e^{-(u^2)} \right\} \sqrt{\frac{t}{KDS_Y}} \quad (17)$$

$$\text{with : } u = \sqrt{\frac{X^2 S_Y}{4KDt}}$$

where: $\operatorname{erfc} = 1 - \operatorname{erf}$ = complementary error function; $q_0 = Q_{IN} (10^6 / 365.25) / (4L)$ [$m^3/(m d)$], with Q_{IN} in Mm^3/a and with $4L$ accounting for 2 drainage sides per recovery system.

The following is assumed: unsteady, one-dimensional flow in an unconfined, semi-infinite aquifer without recharge, above an impervious base. The sudden start of pumping is in reality a sudden change from discharge with steady recharge to discharge without any recharge.

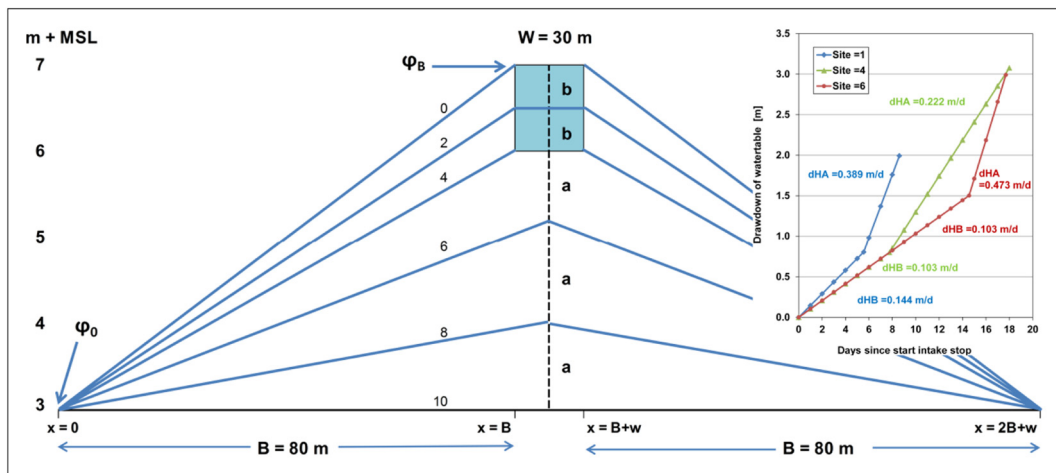


Fig. 6. Calculation scheme of the water table drawdown in the dynamic reservoir during constant abstraction, neglecting the drawdown at the abstraction points at 80 m distance. In this fictional example, 10 days of abstraction lead to emptying of the basin in 4 days and depletion of the aquifer in 6 days. 0–10 = day number after start of intake stop; $a = 2dH_A$ (Eq. (13)); $b = 2dH_B$ (Eq. (12)). Total drawdown (ΔH) = 4 m, and original water table at 3 m above mean sea level. In the inset: the calculated drawdown for sites 1, 4 and 6 in center of the (schematized) basin, with data input in Fig. 4.

The further drawdown with continued pumping after emptying the dynamic reservoir, is thus approximated by Edelman's solution (Eq. (17)). The principle of superposition was applied to Edelman's solution to include the drawdown generated by the 2 parallel drainage systems on each side of the basin (as in Figs. 4 and 6). An example is presented in Fig. 7, showing the calculated drawdown during an intake stop of 100 days, for a schematized cross section over a BAR system with bilateral recovery.

5.2. Declining pressures in the semiconfined aquifer(s)

If water is to be pumped from the deep dune reservoir, water pressure will drop over time, in addition to a decline induced by emptying the dynamic reservoir. This may lead to problems with suction pumps (to be replaced by underwater pumps on some sites) and salinization.

For dune catchment area 1 + 2, the relation between measured hydraulic heads in 3 aquifers and the total annual groundwater abstraction from aquifer 2 (Q) in the period 1915–2011 is shown in Fig. 8. The underlying time series includes the period prior to the commencement of managed aquifer recharge in 1957, showing an increasing Q from the deep reservoir, and the period thereafter with strongly reduced pumping. The plot shows the impact of long lasting, high abstraction rates on hydraulic heads in the phreatic, semiconfined second and semiconfined third aquifer (all fresh).

The drawdown is strongest in the pumped aquifer, as expected. Extrapolation of the Q -H (hydraulic head) relations as given in Fig. 8 yields the following hydraulic heads during a multi-annual intake stop while pumping 41.8 Mm^3/year (sites 1 + 2; Fig. 4) from aquifer 2: -1.5 , -4.9 and -2.6 m ASL in aquifers 1–3 respectively. Such low levels are undesired (especially in the phreatic aquifer for ecological reasons), so that only a fraction of the water demand can be covered by the deep dune reservoir during a multi-annual, full intake stop. About 20 Mm^3/year seems feasible at most, with the remaining part to be covered by pumping from other (dune) areas. In this particular case, water utility PWN can indeed abstract deep dune groundwater from 2 neighboring large dune catchment areas (without MAR), however, during an emergency only.

A nearly equal drawdown relation with: the total annual groundwater abstraction from aquifer 2 was observed by Stuyfzand (1993) on site 4, where deep abstraction started in 1901, increased to 20 Mm^3/year in 1956, and abruptly declined with the onset of MAR in 1957 (Fig. 9). Contrary to site 1 + 2 was the higher mean position of the water table in the phreatic aquifer (3 m ASL), which is explained by the higher resistance of the upper aquitard (Table 3). This higher resistance was not reflected in a greater drawdown in the deep reservoir, because the water

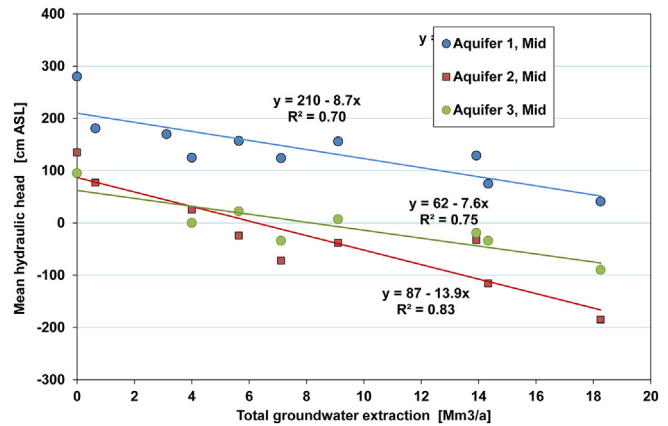


Fig. 8. Relation between the total annual groundwater abstraction from aquifer 2 and hydraulic head in 3 aquifers in the central parts of dune catchment area 1 + 2, in the period 1915–2011. The red years are prior to the start of MAR (in 1957) and show an abstraction increase. The black years (after 1957) show a decreasing abstraction trend.

catchment area at site 4 is larger, and the resistance of the second aquitard is smaller (Table 3).

An important conclusion is, that a multi-annual, full intake stop can be covered on site 1 + 2 by pumping for ~6 years from the deep reservoir without risks of salinization (Table 3), but this may result in unacceptably low hydraulic heads especially in the phreatic aquifer, causing ecological damage.

5.3. Salinization and upconing

Prolonged pumping from the deep dune reservoir will not only lead to a decline of hydraulic heads but also to salinization, as demonstrated by historical records of the overexploitation of the fresh dune water lenses along the Dutch coast (Engelen and Roebert, 1974; Stuyfzand, 1989b, 1993; Olsthoorn and Schaars, 1998).

An example is shown in Fig. 9, displaying a rise of the fresh-salt interface along a cross section over site 4, from the North Sea to reclaimed lake Haarlemmermeer, in the period 1850–1956. This rise was most pronounced in the westernmost parts where lateral seawater intrusion is observed and where upconing was facilitated by a much lower resistance of the deep aquitard. In 1981, 24 years after the start of managed aquifer recharge with a simultaneous strong reduction of the deep pumping, the freshwater expanded again, mainly in that western area where the deep aquitard is less resistant.

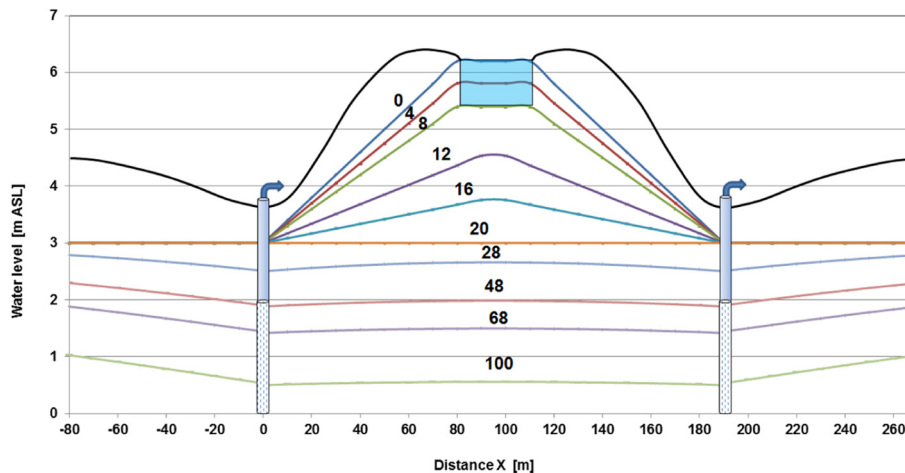


Fig. 7. Calculated drawdown of the water table as function of time (day numbers) since start of intake interruption, for the schematized situation depicted in Figs. 4 and 6. A constant background water table elevation of 3 m ASL and no recharge by rainfall are assumed. Parameters as for site 4 (data in Fig. 4).

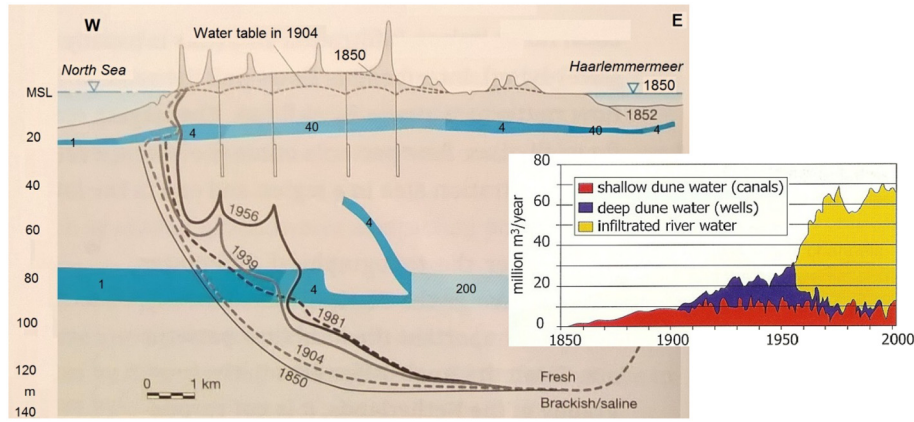


Fig. 9. Left panel: changes in the fresh-salt water interface (8250 mg Cl/L) in a cross section over site 4. Numbers 1–200 indicate resistance of aquitard to vertical flow [years]. Right panel: annual groundwater withdrawal from site 4, with distinction between the 3 contributing water sources. From Stuyfzand (1993).

The upward rise of the fresh-salt interface (v_{UP} ; m/a) can be roughly approximated by the following empirical equation based on historical records from sites 1–6:

$$v_{UP} = a (Q_{DEEP} - c) \quad (18)$$

where: Q = total annual abstraction from the deep reservoir [Mm^3/a]; a = vulnerability factor of the site to salinization, value 0.05–0.2 [–]; c = volume that can be pumped from the deep reservoir without causing salinization [Mm^3/a].

With $a = 0.084$ and $c = 4$ for site 1 + 2, and $a = 0.11$ and $c = 4$ for site 4, the following upward rise is calculated during a multi-annual, full intake stop while pumping from aquifer 2: 3.2 m/a for site 1 + 2 with $Q_{DEEP} = 41.8 Mm^3/a$, and 5.6 m/a for site 4 with $Q_{DEEP} = 55 Mm^3/a$. These numbers are in line with the maximum bridging period from the deep reservoir as given in Table 3, but there are significant spatial variations in salinization risk which need consideration. Abstraction should be maximized from the less vulnerable wells, which are situated according to Fig. 9 in the eastern parts of site 4.

A rational distribution of pumping rates over wells could be based on the upcoming risk of individual wells or well clusters. The risk of upcoming can be estimated by comparing the Safe Depth to the Brackish Interface (SDBI), which is based on Eqs. (8)–(11) and the parameters shown in Fig. 5:

$$SDBI = Z_2 - Z_0 - D_{SAFE} \quad (19)$$

A high value means a low risk of upcoming.

5.4. Rainwater lenses and flow-through dune lakes

In between basin banks and recovery, rainwater lenses may form on top of laterally migrating, infiltrated surface water (Fig. 10A and 11). They are omnipresent, rain fed and therefore often less mineralized and less eutrophic than the fluvial or lacustrine infiltration waters. They may reach a significant thickness only if given a long travel time, and normally show a thin transition zone with the underlying infiltrated surface water. The thickness of these lenses and their transition zone can be predicted with a closed form analytical solution (Stuyfzand, 1993; Stuyfzand, 2016). Rainwater lenses can be important for the survival of rare plant species in wet dune valleys adjacent to recharge basins, by preventing fast growing phreatophytes such as reeds to reach the more eutrophic water and then overshadow rare plant species (Van Dijk, 1989).

The drawdown of water tables during a total intake stop may disturb rainwater lenses by first forcing them downward into zones previously occupied by more eutrophic surface water and subsequently forcing

them back upward when water levels are restored. The short detention in the deeper parts of the aquifer may lead to the uptake of nutrients PO_4 , NH_4 and K , that had been adsorbed during the steady flushing with eutrophic surface water. In addition, the enhanced vertical movement of fresh water lenses will lead to more mixing with the underlying surface water. As a result, the raised nutrient concentrations may stimulate fast growing plants to take over. It is therefore imperative to prevent the decline of water tables in basins where neighboring rainwater lenses are vital. This is feasible only in case of a partial intake stop, during which the reduced volume of infiltration water is spread in the most vulnerable basins only.

Flow-through dune lakes are abundant in the seaward parts of sites 5 and 6 (Fig. 10B and 11), where recharge basins lack a recovery system on the seaward side. This means that a certain loss of infiltrated surface water is allowed to sustain wet dune valleys and flow-through dune lakes, for ecological (EU Natura 2000) reasons. A long-term full intake interruption of these westernmost recharge basins could be detrimental and should therefore be avoided.

5.5. Entrapped air when refilling the basins

It is of paramount importance to refill the recharge basins as fast as possible, in order to be prepared for next intake interruptions and to restore water tables for ecological reasons. This refilling process is, however, sometimes hampered by air entrapment, as observed by e.g. Muckel (1953) in the USA, Lindenberg (1960) on site 5, and by Heilweil et al. (2004) Heilweil et al. (2009) in Utah USA. The infiltration rate (V_{IN}) during basin recharge displays a typical pattern over time (Fig. 12), with an initial decrease caused by dispersion and swelling of soil particles after rewetting, a subsequent increase by dissolution of entrapped air in passing water, and an exponential decrease due to mud formation at the soil-water interface. Entrapped air may strongly reduce V_{IN} for a long time, when aquifer permeability is low (Heilweil et al., 2009). Gas preferentially clogs the larger pore spaces, and may reduce permeability by a factor of 5–10 depending on K_h (10 for 0.7 m/d, 5 for ≥ 100 m/d; Heilweil and Watt, 2011).

Entrapped air can be noticed by evading gas bubbles when piercing the basin floor with a stick, by a disproportional rise of the groundwater table (similar to the so-called Lisse effect; Thal Larsen, 1931; Wang et al., 1998; Weeks, 2002), and by measuring dissolved oxygen or total dissolved gas pressure. Lindenberg (1960) noticed that entrapped air could more easily escape along the basin banks than in the central parts. This spatial pattern is explained by 3 factors: a simple way out along the banks, the 3–10 times higher infiltration rate along the basin banks compared to the central parts (Huisman and Olsthoorn, 1983) leading to a higher gas dissolution rate, and less clogging of the basin

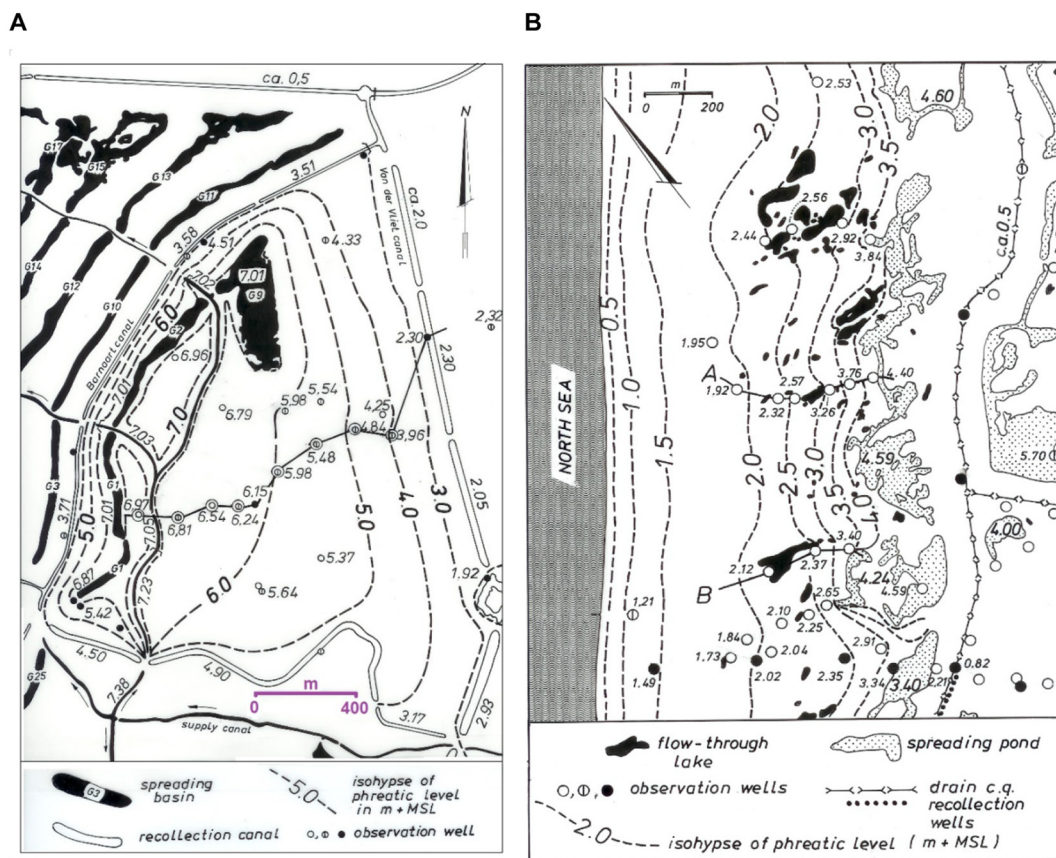


Fig. 10. A: Fragment of site 4 with long flow distances between recharge and discharge allowing rainwater lenses to grow to a thickness up to 6 m (as in Fig. 11A). B: Fragment of site 6 with many flow-through dune lakes in between recharge ponds and discharge into North Sea (as in Fig. 11B). From Stuyfzand (1993).

banks. The presence of a clogging layer is very essential for air entrapment (and the Lisse effect) in coarser grained sediments such as dune sand, as demonstrated by De Zanger (1981). The lack of a clogging layer facilitates the upward escape of gas bubbles as can be seen along the beach when the high tide starts to inundate the lower parts of the beach.

Column tests and many high frequency loggings of water levels in and adjacent to basins on sites 4 and 6 (Cirkel and van der Schans, 2017) revealed that air entrapment can hardly be noticed in basins where muds barely accumulated. This situation contrasts with the period 1957–1974 when pretreatment was much less intensive and muds rapidly formed (Stuyfzand and Mosch, 2002), and air entrapment was noticed to delay the infiltration process for several weeks (Lindenbergh, 1960). It is therefore advised to take advantage of full intake stops to remove mud from the basin floor if significant.

6. Hydrochemical effects

The major water quality effects of an intake stop during which basins and muds dry up, consist of the following: (i) more intense changes in the spreading basins; (ii) oxidation of muds and aquifer; (iii) changes in the removal of organic micropollutants (OMPs); (iv) changes in the contribution of 4–5 water types to the mixed raw water from the recovery system; and (v) a temporarily reduced efficiency of those pre- and post-treatment steps that involve microbiological catalysis. The first 4 effects are discussed below.

6.1. More intense changes in the spreading basins

Lack of recharge leads to a decreasing water depth, an increase of the detention time, and relatively more intense contact with basin muds.

During the refilling stage the average depth remains smaller and the average detention time longer (especially if air is entrapped). These changes conduce to more volatilization, photochemical oxidation, aeration, uptake of nutrients by algae, and biogenic hardness reduction (Stuyfzand, 1989a, 1995a).

As an example, the calculated removal rate (confirmed by field data, not displayed) is shown for chloroform (a by-product of chlorination for water transport) as function of basin depth and detention time in Fig. 13. The removal is clearly increased during the emptying and refilling stage of a recharge basin. The calculation follows the method of Southworth (1979).

6.2. Oxidation of muds and aquifer

The normal spatial distribution of redox zones in dune infiltration systems is presented in Fig. 14, prior to an intake stop with much mud accumulated on the basin floor, and after mud removal and refilling the basin. Most water infiltrates via the relatively clean basin banks and thereby remains (sub)oxic (containing O_2 and NO_3), as long as it is flowing through the upper sand layer and no muds are depositing on the banks. The water becomes anoxic, i.e. Fe-reducing ($O_2 = NO_3 = 0$, SO_4 stable), when passing mud on the banks, sandy clay or clay layers at greater depth, or deeply anoxic (SO_4 -reducing or even methanogenic) where thicker mud at the basin floor or a peat layer is passed.

After falling dry, the muds and underlying unsaturated zone are oxygenated, which oxidizes organic matter and, if present, iron sulfides. When water levels are restored, with or without mud removal, the oxidation products such as SO_4 from iron sulfides and PO_4 from soil organic matter are transported through the aquifer to the recovery system.

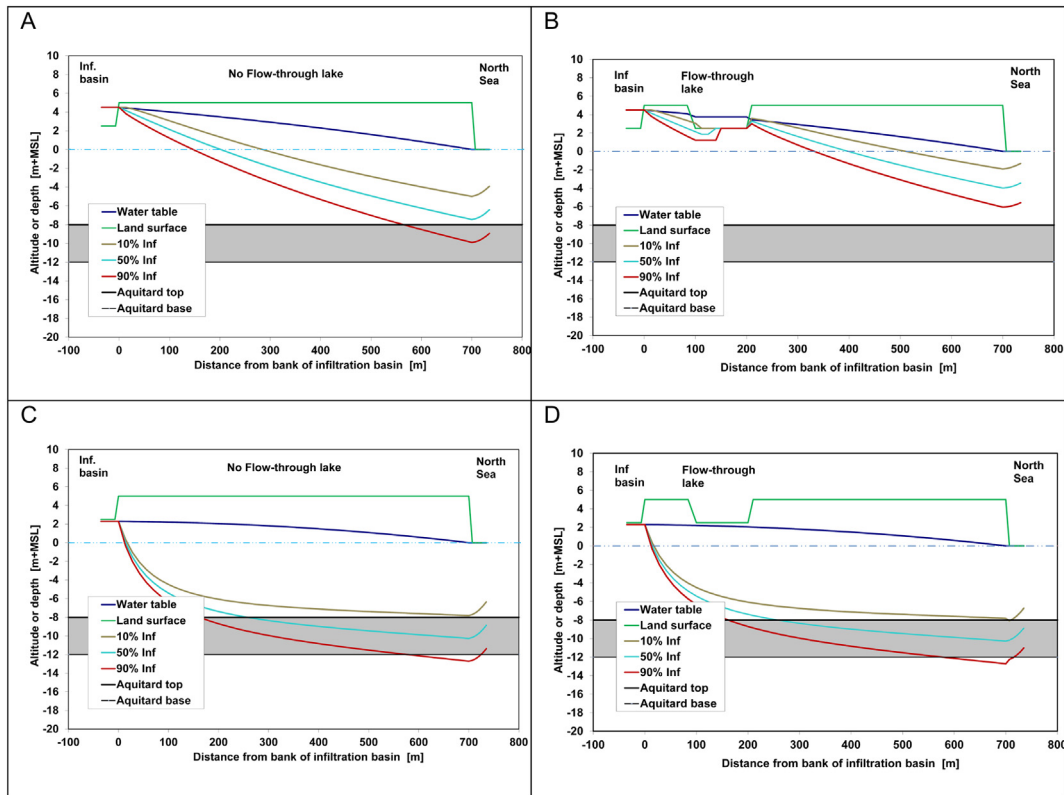


Fig. 11. Calculated position of the water table and rainwater lens (incl. 10–90% mixing zone) with and without a flow-through lake in between recharge basin and the draining North Sea, during recharge operation (A, B) and during a prolonged intake stop (C, D). Important parameters: $R = 0.0012 \text{ m/d}$; $C_1 = 5000 \text{ d}$; $K_h = 12 \text{ m/d}$; $n = 0.38$; $\alpha_T = 0.0025 \text{ m}$. Note that North Sea in reality is on the left hand side (west).

The hydrochemical effects of oxidation strongly depend on the amount of mud accumulated, whether or not mud is removal, and the resulting distribution of redox zones. In case of little mud, the main reaction consists of the oxidation of soil organic matter, from which especially PO_4 is mobilized and transported, with retardation by sorption (Stuyfzand, 1995b; Stuyfzand and Koerselman, 1995), to the recovery (Fig. 15A). During various field studies no effects were observed on e.g. CO_2 , HCO_3^- , Ca, NO_3^- , NH_4^+ , DOC, color, trace elements, OMPs and pathogens (Stuyfzand, 1986; Visser and Fritz, 2003). CO_2 was thus lost to the atmosphere during the mineralization process under dry conditions, because otherwise it would have dissolved

calcium carbonate which is abundant in the aquifer. The current situation shows a strongly reduced presence of mud, thanks to a much lower mud formation potential of the infiltration waters, due to sanitation of the fluvial basins of the Rhine and Meuse Rivers, and an intensified pretreatment since 1974.

In case of much mud, iron sulfides (FeS and FeS_2) form in the deeply anoxic sands below the mud. Their oxidation led to a strong mobilization peak of SO_4 , Ca, Mg and K on site 5 (Fig. 15B). The Ca peak is explained by the acidifying pyrite oxidation and acid buffering by calcite dissolution. The Mg and K peak (not shown) coincides with the Ca and SO_4 peak, which is explained by cation exchange for Ca.

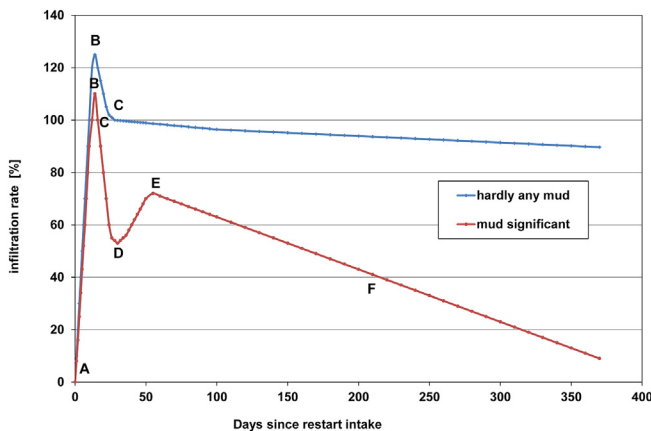


Fig. 12. Typical trend of infiltration rate in a recharge basin, during a period with much mud and hardly any mud. AB = basin (re)filling stage; BC = swelling of soil particles by (re)wetting; CD = increasing effect of air entrapment; DE = entrapped air slowly dissolving in infiltration water; EF = normal decline when significant mud accumulation; F = mud removal desired.

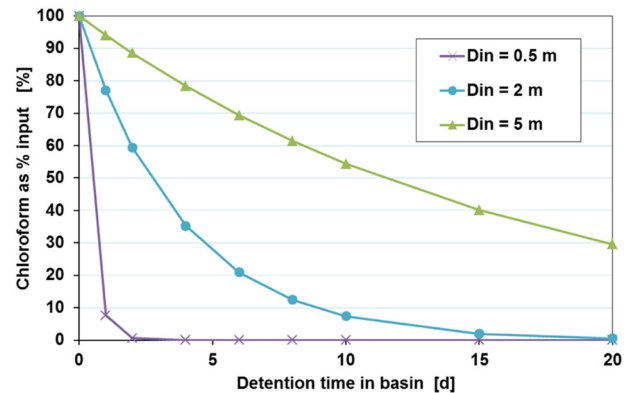


Fig. 13. An intake interruption prolongs detention times in a recharge basin and reduces the water depth during the emptying and refilling stage. That lowers the concentration of volatile and photochemically degradable organic micropollutants such as chloroform. Calculated with the model of Southworth (1979) with wind speed = 5 m/s, flow velocity of water 0.025 m/s, water temperature = 17 °C. D_{in} = water depth.

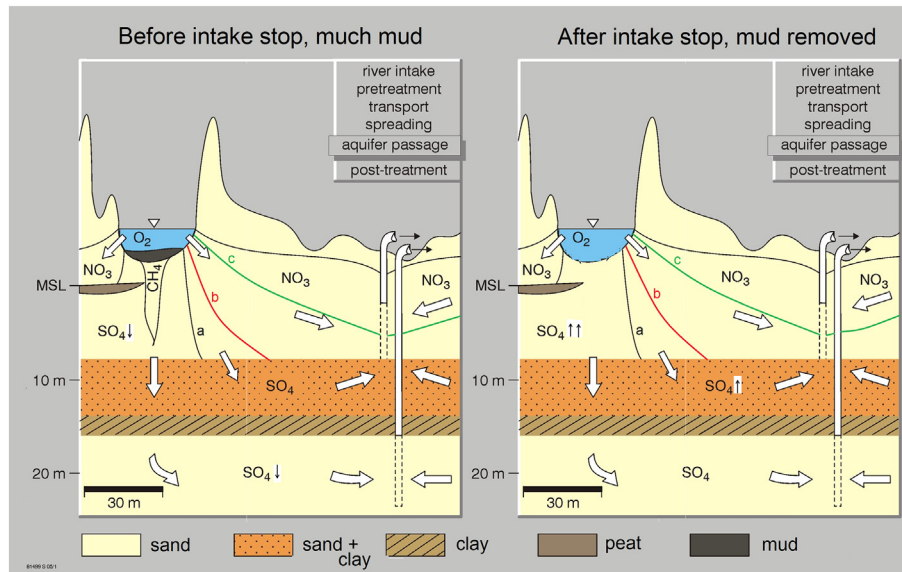


Fig. 14. Spatial distribution of redox zones in dune infiltration systems, before and after a full intake interruption during which mud was removed. O_2 = O_2 saturated (oxic); NO_3 = O_2 -reducing, NO_3 stable to NO_3 reducing (suboxic); SO_4 = fully denitrified, SO_4 stable (anoxic); $SO_4\uparrow$ = anoxic + some SO_4 added by pyrite oxidation; $SO_4\uparrow\uparrow$ = anoxic + much SO_4 added by pyrite oxidation; $SO_4\downarrow$ = deeply anoxic, SO_4 reducing; CH_4 = methanogenic (deeply anoxic, SO_4 largely reduced). a, b, c = NO_3/SO_4 interface without, with some and much mud on basin banks, respectively.

6.3. Changes in removal of organic micropollutants (OMPs) during aquifer passage

Changes in the removal of OMPs may already start in the recharge basins as shown in Fig. 13. Probably more important are the changes in their (bio)degradation rate during aquifer passage, by modifications of redox conditions through oxygenation and mud removal. Lack of dedicated observations force us to do some preliminary calculations with a simple transport equation:

$$C_t = \sigma C_{IN} 2^{\left(\frac{-R_{OMP}t}{T_{1/2}}\right)} \quad (20)$$

where: C_t = concentration of OMP in aquifer at time $t = t$ [% or $\mu\text{g/L}$]; C_{IN} = constant concentration of OMP in input prior to infiltration [100% or $\mu\text{g/L}$]; σ = dissolved fraction (not bound to suspended matter) [–]; R_{OMP} = retardation factor of OMP [–]; t = time since infiltration [d]; $T_{1/2}$ = half-life of OMP [d].

Retardation of OMPs due to hydrophobic sorption is calculated using the classical approach (e.g. Appelo and Postma, 2005), however with addition of the effects of (i) DOC-binding according to Kan and Tomson (1990) and (ii) OMP ionization (dissociation) according to Schellenberg et al. (1984):

$$R_{OMP} = 1 + \frac{f_{G \rightarrow L} f_{OC} f_{ND} K_{OC}}{1 + f_B \text{DOC} 10^{-6} f_{ND} K_{OC}} \quad (21)$$

$$\text{with: } f_{ND} = \frac{1}{1 + 10^{(pH - pK_a)}} \quad (22)$$

where: K_{OC} = distribution coefficient of OMP between organic carbon and water [L/kg C]; f_{OC} = weight fraction of organic carbon in aquifer matrix [–]; f_{ND} = OMP fraction Not Dissociated, thus nonionic and thereby sorbable [–]; $pK_a = -\log K_a$, with K_a = first dissociation constant of acid H-OMP; f_B = fraction (~0.2) of the binding sites supplied by DOC that bind the OMP and prevent it to sorb to the aquifer; $f_{G \rightarrow L}$

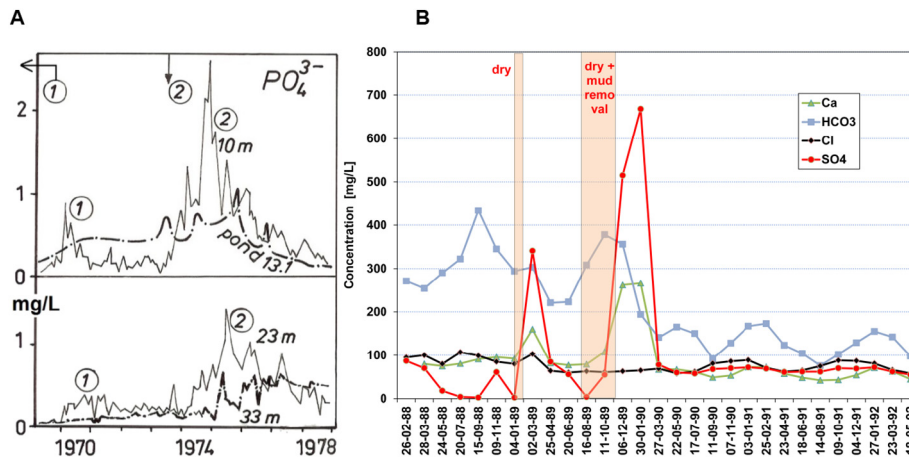


Fig. 15. Effects of an intake stop with mud removal from the basin, in case of little (A) and much (B) mud formation. A: mud was removed from pond 13.1 (site 6) in 1968 and 1973 (arrow). PO_4 measurements in monitoring wells at 10, 23 and 33 m distance from bank. B: mud was removed from pond 21 (site 5) in 1989, measurements in monitoring well Ef2 at 10 m distance from bank.

$= \rho_s (1-n)/n$ with ρ_s = density of solids porous medium [kg/L], and n = porosity [L/L].

In this analytical approach, it is assumed that OMP biodegradation is first order, sorption is solely to organic matter, adsorption and desorption rates are equal; and the degradation rates in solute and adsorbed phase are equal too. In addition, it is assumed that $\sigma = 1$ and $C_{IN} = 100\%$, that the redox conditions change in the upper sand aquifer only, from anoxic into (sub)oxic, and that the physico-chemical constants of 10 selected OMPs (Table 4) remain constant over time. Values of $T_{1/2}$ were taken from various dune infiltration studies among which Stuyfzand et al. (2007b).

The dune infiltration system for which the calculations are done, is schematized as follows. The water recovered is composed of 3 components: 70% is infiltrated surface water from the upper sandy aquifer after 70 m subhorizontal flow in 70 days, 20% is infiltrated surface water from the aquitard and deeper aquifer after 12 m downward flow in 60 days and 70 m subhorizontal flow in 300 days, and 10% is ambient dune groundwater with OMP concentrations equal to zero. The 100% input of the OMP cocktail is assumed to have started 60 years ago and to have remained constant. Important characteristics of the infiltrated surface water, the upper aquifer and aquitard + lower aquifer are given in Table 4.

Then there are 3 cases. Case 1 is without intake stop, the whole aquifer remaining anoxic. Case 2 is with an intake stop, the whole upper aquifer remaining anoxic except for all water that passed the clean basin banks in the upper sand aquifer and thereby became (sub)oxic. And case 3 is also with an intake stop, after which the whole upper sand aquifer became (sub)oxic, also below the basin floor (aquitard and deeper aquifer remaining anoxic). Case 3 therefore shows a major change for the water passing through the aquitard and deeper aquifer, because during downward transport in the upper sand aquifer (assumed 12 m in 60 days) conditions are (sub)oxic now after which the conditions remain anoxic for the remaining 300 days.

The results of calculation in Table 4 demonstrate that an intake interruption including mud removal may have the following consequences for OMP behavior:

- No consequences for persistent OMPs such as MTBE, and for OMPs with a very high retardation (and silt-bound percentage) such as benzo(a)pyrene. MTBE losses in the basin are probably greater.
- Along subhorizontal flowlines in the upper sand aquifer, (bio)degradation significantly changes only if this aquifer was anoxic before, as happened on e.g. site 5 prior to 1989 (Fig. 15B). In that situation (case 2 in Table 4), various OMPs are biodegraded more ($T_{1/2}$ short for (sub)oxic, long for anoxic conditions) and others less ($T_{1/2}$ long for (sub)oxic, short for anoxic conditions). This situation is becoming

rare, however, because quality of the pretreated infiltration waters improved significantly and old mud deposits, that formed when infiltration water was more polluted, have largely been removed.

- Along deeper flowlines passing through the aquitard and deeper aquifer, (bio)degradation significantly improves if (sub)oxic conditions arise over the first 12 m of downward flow in 60 days, followed by anoxic conditions. In this situation (case 3 in Table 4), the contribution of these deep flowlines to the water recovered is, however, limited to 20%, which reduces the average impact on the recovered water to ~10% more removal only. This is the most likely situation for most dune infiltration sites today.

6.4. Changes in contribution of 4–5 water types to recovered water composition

During a prolonged, full intake interruption recovered water quality will significantly change, once the dynamic reservoir has been emptied and water demands have to be completely covered by pumping from the deep reservoir. In that case, infiltrated surface water and some shallow dune groundwater from the phreatic aquifer are replaced, in the mixed output, by infiltrated surface water and deep dune groundwater from the deep aquifer, and if salinization would occur, also some brackish groundwater from the deep aquifer could be admixed.

As an example, the composition of several water types is presented in Table 5 for site 4, together with the calculated quality of the mixed, recovered water prior to post-treatment, before the intake stop (Mix-1) and during later phases of the intake stop when the dynamic reservoir has been emptied (Mix-2). The table also lists the quality of the infiltration water (pretreated Rhine riverwater), and some of the quality requirements for infiltration water (IB, 1993) and drinking water (EU 2012).

We conclude that the shift from Mix-1 to Mix-2, when pumping from the deep reservoir becomes necessary, leads to a significant change in water quality, from dominated by shallow infiltrated Rhine water to dominated by deep dune groundwater. Consequently concentrations of highly soluble salts (Cl, SO_4), oxidants (O_2 and NO_3), various trace elements (Ba, Cd, Cu, Ni, Pb) and practically all OMPs decrease, whereas concentrations increase for constituents that dissolve from the aquifer matrix, notably Ca, HCO_3 , SiO_2 , Fe, Mn, NH_4 and PO_4 .

These changes may cause eutrophication of surface water if the recovery system also consists of open canals (site 4 only). On all sites, they may impact on the post-treatment which is fine-tuned to dominance of recovered surface water (not deep dune groundwater).

Table 4

Calculated effects of aquifer passage by surface water with 100% input of a cocktail of 10 organic micropollutants (OMPs), with and without redox change due to an intake stop with basin drying and mud removal. %OMP in output = OMP concentration in recovered water, as percentage of input. Case 1 = without intake stop, whole aquifer remains anoxic; Case 2 = with intake stop, whole aquifer remains anoxic except for water that passed the (sub)oxic basin banks in the upper sand aquifer; Case 3 = with intake stop, whole upper sand aquifer becomes (sub)oxic, aquitard remains anoxic. R_{OMP} = retardation factor for OMP in upper sandy aquifer and clayey aquitard + lower aquifer.

Org. micropollutant	K_{OC}	pK_a	$T_{1/2}$ [d]		R_{OMP}		%OMP in output		
			Suboxic	Anoxic	Upper	Lower	Case 1	Case 2	Case 3
1,2-Dichloorethene	150		28	720	1.2	1.8	75.5	19.5	10.4
1,2-Dichlorobenzene	309		180	10^{99}	1.5	2.7	90.0	67.3	61.6
Amidotrizoic acid	23		10^{99}	40	1.0	1.1	20.0	70.0	70.1
Benzo(a)pyrene	$2.51 \cdot 10^6$		530	2120	1370	5392	0.0	0.0	0.0
Dinoseb	4140	5	123	10	1.0	1.1	0.5	47.0	47.0
gamma-HCH	1995		240	7	3.9	11.7	0.0	31.6	31.6
HCB	39,811		969	8356	58.2	210	50.0	3.9	3.8
MTBE	12		10^{99}	10^{99}	1.0	1.1	90.0	90.0	90.0
Phenazone	2		2	1460	1.0	1.0	84.5	16.8	0.0
Toluene	204		4	56	1.3	2.1	22.7	0.0	0.0
Average 10 OMPs							43.3	34.6	31.4

Upper aquifer: $f_{OC} = 0.0003$, $n = 0.35$; aquitard + lower aquifer: $f_{OC} = 0.001$, $n = 0.33$.

Infiltrated water in upper aquifer / aquitard + lower aquifer: pH = 7.7 / 7.45; DOC = 3.4 / 3.0 mg/L.

Table 5
Overview of mean water quality anno 2007 in 8 compartments of site 4, with norms for water to infiltrate (MPC input according to IB, 1993) and for drinking water (MPC dr. water according to EU 2012). Rhine 2007 = pretreated Rhine water from intake B (Fig. 1); Inf-1, Inf-2 = infiltrated Rhine riverwater recovered from the dynamic and deep reservoir, respectively; Dune-1, Dune-2, Dune-3 = ambient dune groundwater pumped from dynamic reservoir, deep reservoir and brackish deep reservoir, respectively; Mix-1, Mix-2 = mixture of infiltrated Rhine riverwater and ambient dune groundwater from the dynamic and deep reservoir, respectively. Numbers in red: >MPC; 0.0 = below minimum detection limit (for NO₃ <0.1 mg/L, for OMPs <0.013 µg/L); 0 = assumed zero.

Parameter		Unit	MPC	Rhine	Inf-1	Inf-2	Dune-1	Dune-2	Dune-3	Mix-1	Mix-2	MPC
			Input	2007	Shallow	Deep	Shallow	Deep	Brackish	Shallow	Deep	Drink.W.
General	Q-dyn. reservoir	%	–	–	97	0	3	0	0	100	0	–
	Q-deep reservoir		–	–	0	15	0	84.9	0.1	0	100	–
	EC 20 °C	µS/cm	–	563	577	844	601	544	594	578	589	1250
	pH		–	7.86	7.70	7.50	7.60	7.40	7.40	7.70	7.41	7–9.5
	Temp	oC	–	13.2	12.7	12.3	11.0	10.5	11.2	12.6	10.8	25
Anions	Cl	mg/L	200	79.5	78.8	146	80.6	47.6	64.5	78.8	62.4	150
	SO ₄		150	57.3	54.8	64.6	24.4	16.8	19.1	53.9	24.0	150
	HCO ₃		–	158	184	249	233	288	288	186	282	>60
	NO ₃		24.8	13.3	4.7	0.0	13.9	0.0	0.0	5.0	0.0	50
	PO ₄ -total		1.22	0.06	0.05	0.99	0.21	1.32	1.32	0.05	1.27	6.1
	F		1.00	0.11	0.11	0.09	0.19	0.13	0.13	0.11	0.12	1.1
	Br		–	0.14	0.15	0.20	0.27	0.16	0.22	0.16	0.17	–
	Na	mg/L	120	40.0	39.0	31.8	44.9	28.2	37.5	39.2	28.7	150
Cations and other	K		–	3.7	3.2	3.4	1.7	3.1	3.4	3.2	3.1	–
	Ca		–	69.2	74.2	154.3	84.0	99.9	100.2	74.5	108.1	–
	Mg		–	10.6	9.8	10.1	5.5	7.9	9.1	9.7	8.2	–
	NH ₄		–	0.003	0.200	1.43	0.1	1.28	1.3	0.20	1.3	0.2
	Fe		–	0.005	1.30	3.3	0.05	3.00	3.0	1.26	3.0	0.2 (tot)
	Mn		–	0.0004	0.20	0.68	0.01	0.10	0.10	0.19	0.19	0.05 (tot)
	SiO ₂		–	5.4	5.6	21.2	7.6	45.9	45.9	5.7	42.2	–
	DOC		–	2.8	1.9	2.0	6.4	2.2	2.2	2.1	2.2	–
	As	µg/L	10	1.9	3.8	1.0	8.3	0.5	0.5	3.9	0.6	10
	B		–	40	36.0	31	19	63	67	35	58	500
Trace Elements	Ba		200	66	54	18	14	13	13	53	14	500
	Cd		0.4	0.11	0.10	0.01	0.09	0.02	0.02	0.10	0.02	5
	Cu		15	4.6	3.2	0.3	1.2	0.4	0.4	3.2	0.4	2000
	Hg		0.05	0.026	0.02	0.001	0.02	0.01	0.01	0.02	0.01	1
	Ni		15	3.0	1.6	0.2	0.8	0.2	0.2	1.6	0.2	20
	Pb		15	3.7	0.9	0.10	0.11	0.06	0.1	0.9	0.1	10
	Zn		65	25	3.3	1.0	4.0	3.5	3.5	3.3	3.1	3000
	1,4-Dioxane	µg/L	–	0.80	0.79	2.0	0	0	0	0.77	0.30	–
	Amidotrizoic acid		–	0.17	0.08	0.0	0	0	0	0.08	0.00	–
	AMPA		–	0.40	0.30	0.0	0	0	0	0.29	0.00	–
Organic micropollutants	Atrazine		0.1	0.02	0.02	0.0	0	0	0	0.01	0.00	0.1
	Bentazone		0.1	0.01	0.01	0.3	0	0	0	0.01	0.05	0.1
	Benzene		–	0.01	0.01	0.0	0	0	0	0.00	0.00	1
	Carbamazepine		–	0.08	0.08	0.0	0	0	0	0.08	0.00	–
	Carbendazim		0.1	0.06	0.05	0.0	0	0	0	0.05	0.00	0.1
	Diglyme		–	0.34	0.34	0.5	0	0	0	0.33	0.08	–
	Diurone		0.1	0.03	0.02	0.0	0	0	0	0.02	0.00	0.1
	EDTA		–	4.00	3.92	2.0	0	0	0	3.80	0.30	–
	Glyphosate		0.1	0.05	0.01	0.0	0	0	0	0.01	0.00	0.1
	Iopamidole		–	0.19	0.11	0.0	0	0	0	0.11	0.00	–
	Isoproturone		0.1	0.08	0.04	0.0	0	0	0	0.04	0.00	0.1
	Lindane		0.1	0.01	0.01	0.0	0	0	0	0.00	0.00	0.1
	MCPA		0.1	0.01	0.01	0.0	0	0	0	0.00	0.00	0.1
	Mecoprop (MCP)		0.1	0.01	0.01	0.1	0	0	0	0.01	0.02	0.1
	MTBE		–	0.03	0.03	0.2	0	0	0	0.03	0.03	–
	PFOS		–	0.02	0.02	0.0	0	0	0	0.02	0.00	–
	Sum pesticides		0.5	0.27	0.16	0.4	0	0	0	0.16	0.06	0.5
	THMs		2	0.10	0.05	0.0	0	0	0	0.05	0.00	25 (90%)

7. Ecological effects

The main ecological effects of intake interruptions are limited to aquatic and wet dune slack environments; dry dune areas are hardly affected. The first effects will be noted in the recharge basins when drying up, because fish, water plants and e.g. larvae of dragon-flies will die soon. Some fish may survive, however, where deep pits have been constructed in selected basins. Subsequently surrounding wet dune valleys and flow-through lakes will dry up, leading to the wilting of phreato-phytic plant communities including rare orchids. These “surrounding” effects have been observed during the long period of overexploitation (1900–1957) prior to MAR (Van Zadelhoff, 1981; Geelen et al., 2016). The disappearance of natural dune wetlands clearly resulted in a very

drastic decline of biodiversity (Westhoff, 1989; Grootjans et al., 1997; Grootjans et al., 2013).

In addition to desiccation, ecological effects of oxidation of wet dune soils and muds in flow-through lakes, as well as the disruption or vertical shifts in position of rainwater lenses (Section 5) are suspected, but they are hard to demonstrate during the relatively short intake or infiltration stops as experienced to date (Aggenbach et al., 2014; Stuyfzand and Runhaar, 2016). Oxidation of dune wetlands due to desiccation can lead to temporary eutrophication by mobilization of nutrients, and to an increased decalcification rate especially by changing seepage into infiltration. The high calcium carbonate content of the dune infiltration sites warrants, however, a prolonged buffering of such acidifying events (Stuyfzand, 1998; Stuyfzand et al., 2007a).

Long-term observations in Dutch dune areas, that experienced a strong desiccation of dune wetlands due to groundwater abstraction, also learn that a structural rewetting of these areas may lead to the return of highly appreciated phreatophytic plant communities including rare orchids (Grootjans et al., 1997). This return may require, however, stimulating measures such as mowing and sod cutting to reduce the nutrient status (which has been affected by excessive atmospheric nitrogen deposition as well; Kooijman et al., 1998).

In order to reduce ecological damage by intake stops, the most vulnerable areas are to be kept wet as long as possible, by recharging the basins with a direct impact on the vulnerable areas as long as possible and by immediately reducing or halting their recovery.

8. How to respond to intake interruptions?

The DSPIR (Driver, Pressures, State changes, Impacts and Responses) causal framework of the European Environmental Agency (Kristensen, 2004) helps to order the triggers and consequences of intake interruptions for ATR systems in a coastal environment, and the potential responses. The result is shown in Fig. 16. The focus here is on the curative and preventive responses.

Curative responses consist of measures that mitigate the effects of (full) intake interruptions. If timely announced through an efficient early warning system and luck (detected far upstream), then the recharge basins can be (re)filled to their maximum level, thus maximizing the dynamic reserves. Whenever possible and necessary, muds should be removed from basins, in order to prevent air entrapment to delay the refill process. The deep reservoir could be refilled by using the deep abstraction wells as infiltration wells, which requires however significant asset investments. Fresh keeper wells (Kooijman et al., 2004) may help to prevent upconing of saline groundwater on vulnerable sites, but need to be constructed. Basins near ecologically vulnerable

areas could be recharged by recirculating part of the water recovered. And if needed, the pumping and water delivery could be reduced, in combination with communication about how to save water. An important bottleneck of rapid (re)filling is the capacity of the pretreatment and transport mains from pretreatment to dune area.

Preventive responses comprise measures that reduce the risk of intake interruptions. The best measure is to sanitize the whole fluvial basin, especially upstream of the intake. The success of sanitation measures has been demonstrated by the significant quality improvements of the Rhine River since the 1970s (LIT). The precautionary principle that “the polluter should pay”, must change into ‘the polluter should pay significantly more’. A second, independent intake has proven already to be an excellent way to prevent full intake stops on all dune infiltration sites (Table 1 and Fig. 3). The last resort is to strengthen the pretreatment system so as to be capable to remove pollutant peaks (done for sites 1 + 2, Table 1).

9. Conclusions

The duration and frequency of intake interruptions for managed aquifer recharge (MAR) systems are expected to increase due to climate change (more droughts and higher peak flows) and continued environmental pollution. These interruptions form a normal problem and even the reason for some MAR types to exist, but for Aquifer Transfer and Recovery (ATR) systems they pose more serious troubles, because ATR systems need to continuously deliver water by recovery after aquifer passage.

In this contribution, we focused on 8 coastal dune infiltration systems in the Western Netherlands, which belong to the ATR type. Their vulnerability to intake stops is further raised, during continued abstraction, by high risks of salt water intrusion, upconing and ecological damage to wet dune slacks with an EU Natura 2000 status, which cannot

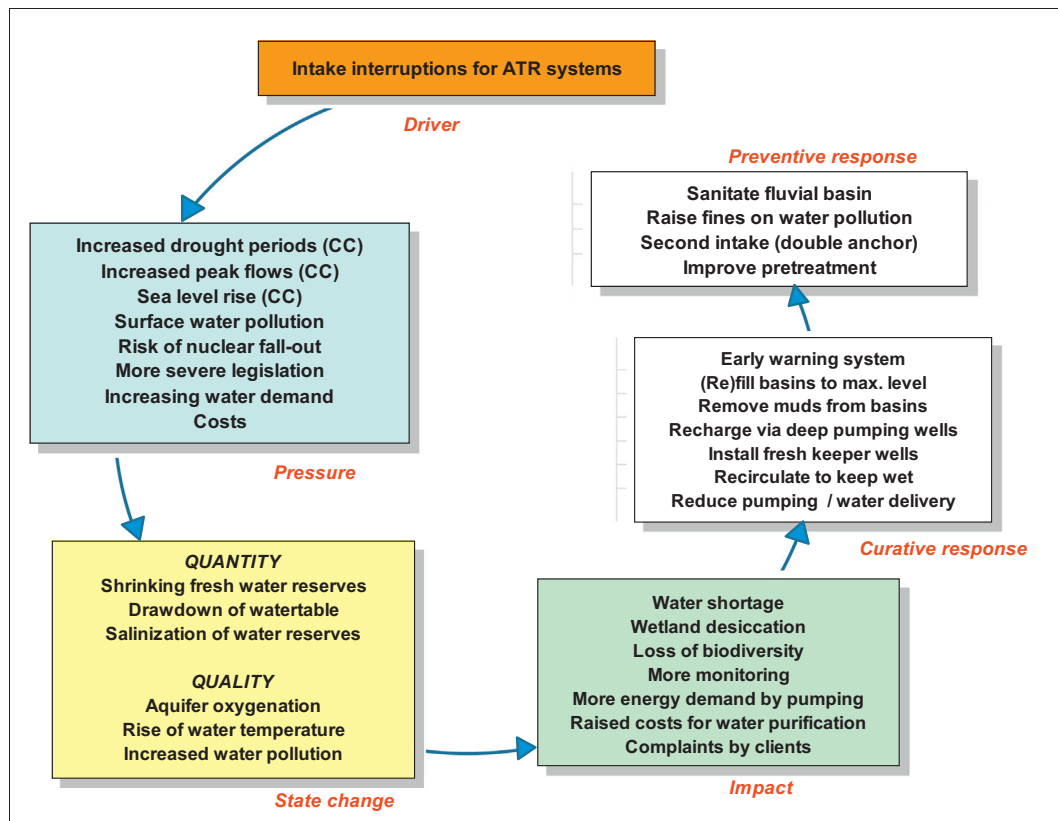


Fig. 16. The DSPIR causal framework of the European Environmental Agency (Kristensen, 2004) implemented to order the triggers and consequences of intake interruptions for ATR systems in a coastal environment, together with potential responses. DSPIR = Driver, Pressures, State changes, Impacts and Responses. CC = climate change.

survive for long without managed aquifer recharge. This susceptibility triggered various measures to raise the security of water delivery by the Dutch dune infiltration systems, such as: internationally coordinated sanitation measures in the fluvial catchment areas of the Rhine River and Meuse River, a second/emergency intake (all 8 sites), a solid pretreatment (all 8 sites) which was further advanced on 2 sites, and a diffuse network of wells to pump fresh groundwater from the semiconfined reservoir if needed during a long, full intake stop.

Calculations and practice demonstrate that short, total intake interruptions (4–20 days for sites 1–8) can be covered by pumping from the so-called dynamic reservoir. This reservoir consists of the recharge basins, shallow recovery systems and the surrounding upper aquifer above 1–3 m above sea level, mainly filled with (infiltrated) surface water. During a longer full intake stop the deep reservoir is to be pumped, which can bridge, according to calculations and historical evidence, a period of several years (5–7 for the larger sites), provided that wells vulnerable to upconing are charged less and that the deep well capacity is sufficient.

The hydrological effects of pumping from the stored water volumes, consist of the drawdown of water tables, disappearance of flow-through dune lakes, reservoir depletion, salt water intrusion and upconing, the disruption of so-called rainwater lenses on top of laterally migrating infiltrated surface water, and entrapped air frustrating a rapid refill of the groundwater reservoir. Analytical solutions are presented to quantify some of these effects; they are (partly) also valid for pumped coastal aquifers without MAR.

Water quality changes within the dynamic reservoir, due to the drawdown of water tables and drying up of basins during intake stops, were more serious prior to 1974–1989 when muds rapidly formed and (deeply) anoxic conditions prevailed. The oxidation of accumulated organic matter and iron sulfides resulted in high peaks of mobilized PO_4 , Ca and SO_4 . In recent years much less muds are forming and such mobilization peaks have not been observed anymore. With a simple analytical model it is calculated that the drying up of basins leads to more removal of volatile organic micropollutants (OMPs) and that expansion of the shallow (sub) oxic environment at the expense of anoxic zones is, on average, conducive to more OMP (bio)degradation.

During prolonged, total intake stops, after emptying the dynamic reservoir, pumping is to be continued with water from the deep reservoir. The resulting shift leads to a significant change in water quality, from dominated by shallow infiltrated river water to dominated by deep dune groundwater. Consequently concentrations of highly soluble salts (Cl, SO_4), oxidants (O_2 and NO_3), various trace elements (Ba, Cd, Cu, Ni, Pb) and practically all OMPs decrease, whereas concentrations increase for constituents that dissolve from the aquifer matrix, notably Ca, HCO_3 , SiO_2 , Fe, Mn, NH_4 and PO_4 .

The main ecological impacts comprise the dying of organisms in recharge ponds and dune lakes, and a decline of biodiversity. In order to reduce ecological damage by intake stops, the most vulnerable areas are to be kept wet as long as possible, by recharging the basins with a direct impact on the vulnerable areas as long as possible and by immediately reducing or halting their recovery.

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References

- Aggenbach, C.J.S., van der Schans, M.L., Sjerps, R.M.A., Zwolsman, J.J.G., 2014. Handelingschema alternatieve inname rivierwater en innamestops duininfiltratie gebaseerd op hydrologische, chemische en ecologische effecten. KWR Report BTO 2014.212(s) (123 pp.).
- Appelo, C.A.J., Postma, D., 2005. *Geochemistry, Groundwater and Pollution*. 2nd edition. Balkema, Rotterdam (649 pp.).
- Aquapedia, 2014. https://aquapedia.waterdiplomacy.org/wiki/index.php?title=Water_Quality_and_Pollution_Control_in_the_Rhine_River_Basin, Accessed date: 8 February 2018.
- Asano, T. (Ed.), 1985. *Artificial Recharge of Groundwater*. Butterworth Publishers, USA (767 pp.).
- Bouwer, E., et al., 2008. Prospects for Managed Underground Storage of Recoverable Water. National Research Council, National Acad. Press, Washington www.nap.edu (337 pp.).
- Bruggeman, G.A., 1999. *Analytical Solutions of Geohydrological Problems*. Elsevier, Amsterdam.
- Chadha, D.K., 2002. State of art of artificial recharge applied on village level schemes in India. Management of Aquifer Recharge and Subsurface Storage; Making Better Use of Our Largest Reservoir. Netherlands National Committee IAH, Publ. No.4, pp. 19–24.
- Cirkel, G., van der Schans, M., 2017. Air entrapment under dune recharge basins after intake interruptions. KWR Report KWR 2017-07? (in Dutch).
- Davis, J.S., Zobrist, J., 1978. The interrelationships among chemical parameters in rivers: analyzing the effect of natural and anthropogenic sources. *Prog. Water Technol.* 10, 65–78.
- De Zanger, F., 1981. Exceptional groundwater level fluctuations at the Dutch Island of Schiermonnikoog. *Hydrol. Res.* 12 (2), 111–118.
- Dillon, P.J., 2005. Future management of aquifer recharge. *Hydrogeol. J.* 13 (1), 313–316.
- Engelen, G.B., Roebert, A.J., 1974. Chemical water types and their distribution in space and time in the Amsterdam dunewater catchment area with artificial recharge. *J. Hydrol.* 21, 339–356.
- Fox, P., et al., 2006. *Advances in Soil Aquifer Treatment for Sustainable Water Reuse*. Awwa Research Foundation, USA (200 pp.).
- Geelen, L.H.W.T., Kamps, P.T.W.J., Olsthoorn, T.N., 2016. From overexploitation to sustainable use, an overview of 160 years of water extraction in the Amsterdam dunes, the Netherlands. *J. Coast. Conserv.* 21:657–668. <https://doi.org/10.1007/s11852-016-0452-x>.
- Grootjans, A.P., Esselink, H., Van Diggelen, R., Hartog, P., Jager, T.D., Van Hees, B., Oude, Muninck, J., 1997. Decline of rare calciphilous dune slack species in relation to decalcification and changes in local hydrological systems. In: Garcia Novo, F., Crawford, R.M.M., Cruz Diaz Barrados, M. (Eds.), *The Ecology and Conservation of European Dunes*. Univ. Sevilla, pp. 59–74.
- Grootjans, A.P., Duilio, B.S., Kooijman, A.M., Bekker, R.M., Aggenbach, C., 2013. Restoration of dune vegetation in the Netherlands. Ch.15. In: Martinez, M.L., et al. (Eds.), *Restoration of Coastal Dunes*. Springer Series on Envir. Management, pp. 235–254.
- Heilweil, V.M., Watt, D.E., 2011. Trench infiltration for managed aquifer recharge to permeable bedrock. *Hydrol. Process.* 25 (1), 141–151.
- Heilweil, V.M., Kip Solomon, D., Perkins, K.S., Ellett, K.M., 2004. Gas-partitioning tracer test to quantify trapped gas during recharge. *Ground Water* 42:589–600. <https://doi.org/10.1111/j.1745-6584.2004.tb02627.x>.
- Heilweil, V.M., Kip Solomon, D., Ortiz, G., 2009. Silt and gas accumulation beneath an artificial recharge spreading basin, SW Utah, USA. *Bol. Geol. Min.* 120 (2), 185–196.
- Huisman, L., Olsthoorn, T.N., 1983. *Artificial Groundwater Recharge*. Pitman Adv. Publ. Program, London (320 pp.).
- IB, 1993. Infiltratiebesluit Bodembescherming, houdende regels met betrekking tot infiltratie van uit oppervlaktewater verkregen water in de bodem. *Staatsblad Koninkrijk Nederlanden* 1993, 233 (14 pp.).
- Jelgersma, S., De Jong, J., Zagwijn, W.H., Van Regteren Altena, J.F., 1970. *The Coastal Dunes of the Western Netherlands: Geology, Vegetational History and Archeology*. Meded. Rijks Geol. Dienst, NS. 21 pp. 93–167.
- Kan, A.T., Tomson, M.B., 1990. Ground water transport of hydrophobic organic compounds in the presence of dissolved organic matter. *Environ. Toxicol. Chem.* 9, 253–263.
- Kooijman, A.M., Dopheide, J.C.R., Sevink, J., Takken, I., Verstraten, J.M., 1998. Nutrient limitations and their implications on the effects of atmospheric deposition in the coastal dunes; lime-poor and lime-rich sites in the Netherlands. *J. Ecol.* 86, 511–526.
- Kooiman, J.W., Stuyfzand, P.J., Maas, C., Kappelhof, J.W., 2004. Pumping brackish groundwater to prepare drinking water and keep salinizing wells fresh: a feasibility study. *Proc. 18th SWIM*, Cartagena, Spain, pp. 625–635.
- Kristensen, 2004. The DPSIR Framework. Paper Presented at the 27–29 September 2004 Workshop on a Comprehensive/detailed Assessment of the Vulnerability of Water Resources to Environmental Change in Africa Using River Basin Approach. UNEP Headquarters, Nairobi, Kenya <http://www.ifremer.fr/dce/content/download/69291/913220/file/DPSIR.pdf> (visited on 14 May 2017).
- Lindenbergh, P.C., 1960. Experiences With Infiltration of Water in the Dune Catchment Area of Leidsche Duinwater Maatschappij (L.D.M.), Verslagen technische bijeenkomsten 13 en 14. versl. meded. comm. hydrol. onderz. T.N.O. No. 5, 's Gravenhage (in Dutch).
- Maliva, R.G., Missimer, T.M., 2010. *Aquifer Storage and Recovery and Managed Aquifer Recharge Using Wells: Planning, Hydrogeology, Design, and Operation*. Schlumberger Water Services, Sugarland, Texas (578 pp.).
- Muckel, D.C., 1953. Research in water spreading. *Trans. Am. Soc. Civ. Eng.* 118, 209–219.
- Nienhuis, P.R., Olsthoorn, T.N., Kamps, P.T.W.J., 2012. Modelling AS(T)R in the Amsterdam water supply dunes to explore options to cope with future intake water shortages caused by climate change. *Proceedings of 22nd Salt Water Intrusion Meeting SWIM 2012*. Armação dos Búzios, pp. 138–139.

- Olsthoorn, T.N., Schaars, F.W., 1998. Variable density groundwater modelling with MODFLOW: example Amsterdam dune water area. In: Poeter, E., et al. (Eds.), Proceedings of the MODFLOW98 Golden Conf. Colorado School of Mines, pp. 579–586.
- Pyne, R.D.G., 2005. *Aquifer Storage Recovery: A Guide to Groundwater Recharge through Wells*. 2nd edition. ASR Press, Gainesville, Florida (608 pp.).
- RIWA-Rijn, 2015. *De kwaliteit van het Rijnwater in 2015*. <http://www.riwa-rijn.org/wp-content/uploads/2016/09/IDF1331-RIWA-jaarrapport-NL-2015-internet-kl.pdf>.
- Schellenberg, K., Leuenberger, C., Schwarzenbach, R.P., 1984. Sorption of chlorinated phenols by natural sediments and aquifer materials. *Environ. Sci. Technol.* 18, 652–657.
- Sjerps, R.M.A., ter Laak, T.L., Zwolsman, J.J.G., 2017. Projected impact of climate change and chemical emissions on the water quality of the European rivers Rhine and Meuse: a drinking water perspective. *Sci. Total Environ.* 601–602, 1682–1694.
- Southworth, G.S., 1979. The role of volatilization in removing PAH from aquatic environments. *Bull. Environ. Contam. Toxicol.* 21, 507–514.
- Stuyfzand, P.J., 1986. Macroparameters and dune infiltration; quality changes of surface water upon artificial recharge in the Dutch coastal dunes. *KIWA-Meded.* 82 (345 pp., in Dutch).
- Stuyfzand, P.J., 1989a. Quality changes of river Rhine and Meuse water upon basin recharge in The Netherlands' coastal dunes: 30 years of experience. In: Johnson, A.I., Finlayson, D.J. (Eds.), *Artificial Recharge of Groundwater*, Proc. Int. Symp. Anaheim USA, 21–28 August 1988. Am. Soc. Civil Eng., New York, pp. 233–245.
- Stuyfzand, P.J., 1989b. Hydrochemical Evidence of Fresh and Salt Water Intrusions in the Coastal Dunes Aquifer System of the Western Netherlands. (Flemish) *Natuurwetensch. Tijdsch.* 70 pp. 9–29.
- Stuyfzand, P.J., 1993. Hydrochemistry and Hydrology of the Coastal Dune Area of the Western Netherlands. Vrije Univ. Amsterdam, KIWA 90-7474101-0 <http://dare.uvu.nl/handle/1871/12716> (Ph.D Thesis, 366 pp.).
- Stuyfzand, P.J., 1995a. The impact of land reclamation on groundwater quality and future drinking water supply in The Netherlands. *Water Sci. Technol.* 31 (1995), 47–57.
- Stuyfzand, P.J., 1995b. Behavior of phosphate in eutrophic surface water upon artificial recharge in the Western Netherlands. In: Johnson, A.I., Pyne, R.G.D. (Eds.), *Artificial Recharge of Groundwater*, II, Proc. SISAR (Second Int. Symp. on Artif. Recharge), Orlando USA, 17–22 July 1994. Am. Soc. Civil Eng., New York, pp. 558–567.
- Stuyfzand, P.J., 1998. Decalcification and acidification of coastal dune sands in the Netherlands. In: Arehart, G.B., Hulston, J.R. (Eds.), *Water-rock Interaction*, Proc. 9th Intern. Symp. on WRI, Taupo New Zealand. Balkema, pp. 79–82.
- Stuyfzand, P.J., 2017. Observations and analytical modeling of freshwater and rainwater lenses in coastal dune systems. *J. Coast. Conserv.* 21 (5):577–593. <https://doi.org/10.1007/s11852-016-0456-6>.
- Stuyfzand, P.J., Koerselman, W., 1995. Natuurherstel in infiltratiegebieden. *Landschap*. 5 pp. 13–28.
- Stuyfzand, P.J., Mosch, M., 2002. Formation and composition of sludges in recharge basins, recollection canals and natural lakes in Amsterdam's Dune Catchment area. In: Dillon, P.J. (Ed.), *Management of Aquifer Recharge for Sustainability*, Proc. 4th Internat. Symp. on Artificial Recharge, Adelaide, Australia, 22–26 Sept. 2002. Balkema, pp. 495–498.
- Stuyfzand, P.J., Runhaar, H., 2016. Naar innamestop-kwetsbaarheidskaarten gebaseerd op hydrologische, hydrochemische en ecologische aspecten. KWR Report KWR-2016.020 (84 pp.).
- Stuyfzand, P.J., Kortleve, M., Olsthoorn, T.N., Rolf, H., 2007a. Accumulation of pollutants and aquifer leaching during artificial recharge compared to a natural recharge system, in the Netherlands. In: Fox, P. (Ed.), *Management of Aquifer Recharge for Sustainability*. Proc. ISMAR-6, 28 Oct–2 Nov 2007, Phoenix AR USA. Acacia Publ. Inc., Phoenix, pp. 174–184.
- Stuyfzand, P.J., Segers, W., Van Rooijen, N., 2007b. Behavior of pharmaceuticals and other emerging pollutants in various artificial recharge systems in the Netherlands. In: Fox, P. (Ed.), *Management of Aquifer Recharge for Sustainability*. Proc. ISMAR-6, 28 Oct–2 Nov 2007, Phoenix AR USA. Acacia Publ. Inc., Phoenix, pp. 231–245.
- Thal Larsen, J.H., 1931. Over den invloed van regenval op den grondwaterstand. *Landbouwk. Tijdschr.* 43, 222–240.
- Todd, D.K., 1959. *Ground Water Hydrology*. Wiley & Sons, New York (336 pp.).
- Van Dijk, H.W.J., 1989. Ecological impact of drinking water production in Dutch coastal dunes. In: Van der Meulen, F., Jungerius, P.D., Visser, J.H. (Eds.), *Perspectives in Coastal Dune Management*. SPB Acad. Publ. B.V., The Hague, The Netherlands, pp. 163–182.
- Van Zadelhoff, F.J., 1981. *Nederlandse kustduinen*; Geobotanie. Pudoc, Wageningen.
- Visser, T., Fritz, J., 2003. Gevolgen slibverwijdering pan 13 in het infiltratiegebied Meyendel van Duinwaterbedrijf Zuid-Holland; Resultatenoverzicht 1988–1993. *DZH-Rapport 20.25.14* (93 pp.).
- Vliet, M., Zwolsman, G.J., 2008. Impact of summer droughts on the water quality of the Meuse River. *J. Hydrol.* 353 (1):1–17. <https://doi.org/10.1016/j.jhydrol.2008.01.001>.
- Wang, Z., Feyen, J., van Genuchten, M.T., Nielsen, D.R., 1998. Air entrapment effects on infiltration rate and flow instability. *Water Resour. Res.* 34 (2):213–222. <https://doi.org/10.1029/97WR02804>.
- Weeks, E.P., 2002. The Lisse effect revisited. *Ground Water* 40:652–656. <https://doi.org/10.1111/j.1745-6584.2002.tb02552.x>.
- Westhoff, V., 1989. Dunes and dune management along the North Sea coast. In: Van der Meulen, F., Jungerius, P.D., Visser, J.H. (Eds.), *Perspectives in Coastal Dune Management*. SPB Acad. Publ. B.V., The Hague, The Netherlands, pp. 41–51.
- Zagwijn, W.H., 1985. An outline of the Quarternary stratigraphy of The Netherlands. *Geol. Mijnb.* 64, 17–24.
- Zwolsman, J.J.G., Van Bokhoven, A.J., 2007. Impact of summer drought on water quality of the Rhine River – a preview of climate change? *Water Sci. Technol.* 56, 45–55.